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Monitoring in the Vadose Zone
A Review of Technical Elements and Methods

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A Review of Technical Elements and Methods

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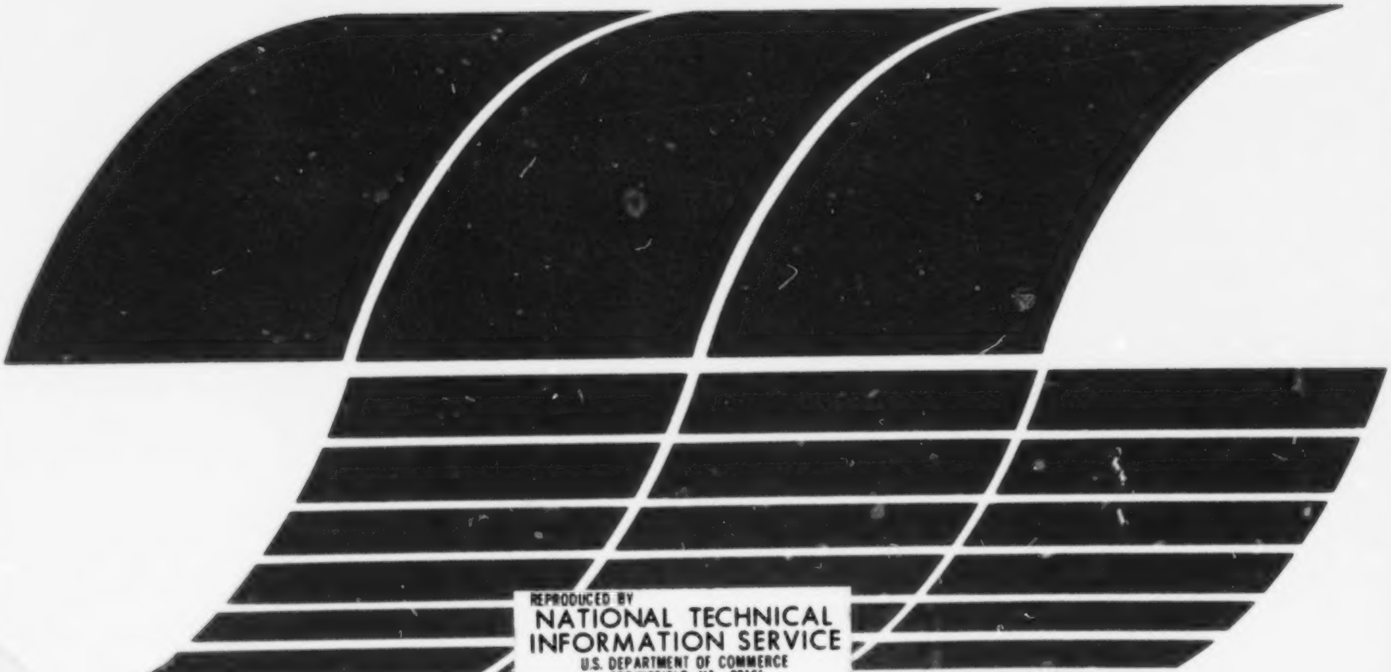

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16. ABSTRACT

This report covers the topics of (1) principles of pollutant movement in the vadose zone (zone of aeration or unsaturated zone), (2) basic chemical reactions of fluids in the zone, (3) state-of-the-art monitoring techniques, and (4) the relative advantages and disadvantages of the different monitoring techniques. Recent intense concern over hazardous waste disposal has indicated the need for instruction on how to monitor in the vadose zone and to identify the potential gains from the limitations of the methods available for monitoring in this zone. This report provides technical information needed by regulating agencies and industrial concerns in dealing with waste disposal problems. In addition, the basis for future research is provided through identification of present monitoring limitations.

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A Review of Technical Elements and Methods

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FOREWORD

Protection of the environment requires effective regulatory actions based on sound technical and scientific data. The data must include the quantitative description and linking of pollutant sources, transport mechanisms, interactions, and resulting effects on man and his environment. Because of the complexities involved, assessment of exposure to specific pollutants in the environment requires a total systems approach that transcends the media of air, water, and land. The Environmental Monitoring Systems Laboratory at Las Vegas contributes to the formation and enhancement of a sound monitoring-data base for exposure assessment through programs designed to:

- Develop and optimize systems and strategies for monitoring pollutants and their impact on the environment
- Demonstrate new monitoring systems and technologies by applying them to fulfill special monitoring needs of the Agency's operating programs.

This report covers the topics of (1) principles of pollutant movement in the vadose zone (zone of aeration or unsaturated zone), (2) basic chemical reactions of fluids in the zone, (3) state-of-the-art monitoring techniques, and (4) the relative advantages and disadvantages of the different monitoring techniques. Recent intense concern over hazardous waste disposal has indicated the need for instruction on how to monitor in the vadose zone and to identify the potential gains from the limitations of the methods available for monitoring in this zone. This report provides technical information needed by regulating agencies and industrial concerns in dealing with waste disposal problems. In addition, the basis for future research is provided through identification of present monitoring limitations.

Further information on this study and the subject of groundwater quality monitoring in general can be obtained by contacting the Environmental Monitoring Systems Laboratory, U.S. Environmental Protection Agency, Las Vegas, Nevada.

Glenn E. Schweitzer
Director
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SUMMARY

The land surface is being utilized increasingly for the disposal of solid and liquid wastes. Concomitantly, there has been an increasing awareness of the need to monitor the movement of pollutants from such wastes into groundwater. Sampling pump discharge offers a direct approach for monitoring groundwater. Unfortunately, by the time pollutants appear in well discharge samples, the groundwater system may have become extensively contaminated. An alternative approach is to monitor within the vadose zone. Water samples from this region may presage the quality of water entering the underlying flow system and hence manifest the ultimate pollution potential. Ameliorative measures could then be initiated before groundwater contamination becomes extensive.

In contrast to the advanced state of the art for sampling from the zone at saturation, vadose zone monitoring is still in its infancy. The discrepancy in technology in part reflects the greater complexity of flow in the vadose zone, compared to saturated zone flow, and the associated problem of obtaining a representative water sample for analysis. The lack of a comprehensive methodology for vadose zone monitoring is a serious impediment to federal and state personnel charged with promulgating or enforcing monitoring regulations for best management practices in agriculture, for hazardous waste disposal programs, and for 208 programs. The majority of these individuals are untrained in the hydraulics of saturated and unsaturated groundwater flow. Therefore, a need exists for a document presenting a comprehensive review of techniques for monitoring water and pollutant movement in the vadose zone.

The purpose of this report is to serve as a basic introduction to the three components of a vadose zone monitoring program. These components are: (1) changes in storage of water in the vadose zone, (2) the flow rate of water in the vadose zone, and (3) the temporal and spatial distribution of pollutants in the vadose zone. A brief technical review of each of these components is presented to set the stage for a review of associated field methods. The principles of these methods are briefly discussed, together with some of the advantages and disadvantages.

Techniques that are reviewed for observing storage changes in the vadose zone at a waste disposal site include the following:

- Monitoring the spatial distribution of water levels in wells to delineate the areal thickness of the vadose zone

- The gamma ray attenuation method to characterize bulk density and water content of vadose zone sediments
- The neutron moderation method for defining the water content distribution in the vadose zone
- Tensiometers for estimating water content at discrete points in the vadose zone
- Electrical resistance blocks for estimating water content at discrete points.

Methods for monitoring water movement (flux) and associated parameters in the vadose zone include:

- Estimating infiltration rates by infiltrometers and test ponds
- Characterizing the quantity of water moving beneath the soil zone using the water balance approach
- Determining the direction of unsaturated water movement and associated hydraulic gradients using tensiometers, psychrometers, and the neutron moderation method
- Measuring the unsaturated flux of water by adapting laboratory techniques to the field, using water content profiles, estimating from suction cup response, and using direct techniques such as flow meters
- Determining saturated flow in perched groundwater zones using piezometers and observation wells
- Outlining techniques for determining the saturated hydraulic conductivity in the soil zone and deeper vadose zone.

Indirect methods for monitoring movement in the vadose zone which are reviewed include the following:

- The four-electrode method for soil salinity
- The EC probe for monitoring soil salinity
- The four-electrode conductivity cell for observing soil salinity
- The earth resistivity approach for delineating pollution plumes.

Techniques for solids sampling in the vadose zone for determination of associated pollutants are reviewed.

The following direct techniques for water sampling during unsaturated flow are described:

- Ceramic-type samplers (suction lysimeters and filter candles)
- Cellulose-acetate hollow-fiber filters
- Membrane filter samplers.

Methods for sampling from shallow perched groundwater zones include:

- Sampling tile drain outflow
- Collection pans and manifolds
- Wells
- Piezometers
- Multilevel samplers
- Groundwater profile samplers.

Sampling from deeper perched groundwater includes:

- Collecting cascading water
- Installing special wells.

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LIST OF ABBREVIATIONS AND SYMBOLS

ABBREVIATIONS

atm	Unit of pressure, the atmosphere, equivalent to 1.013×10^6 dyne/cm^2, or 1.013 bars
bar	Unit of pressure, equivalent to 10^6 dyne/cm^2, or 0.987 atmosphere
CEC	cation exchange capacity
cm	centimeters
EC	electrical conductivity
EC_a	electrical conductivity of a saline soil
EC_e	conductivity of saturation extract
EC_s	surface conductance
EC_w	electrical conductivity of water or soil solution
Eh	redox potential
ESP	exchangeable sodium percentage
ET	evapotranspiration
I	irrigation
I_t	cumulative infiltration
meq	milliequivalents
mmhos/cm	millimhos per centimeter
PET	potential evapotranspiration component of water balance equation
pH_c	pH adjusted for calcium, magnesium, carbonate bicarbonate equilibria
PPT	precipitation plus irrigation components of water balance equation
R	rainfall
RO	runoff component of water balance equation
SAR	sodium adsorption ratio
SP	saturation percentage

SYMBOLS

A	Cross-sectional area of aquifer normal to flow; geometric factor in piezometer equation; factor in Philip's infiltration equation
a	Empirical factor relating k and θ in Nielsen-Biggar flux equation; distance between electrodes in four-probe salinity method
D_b	Bulk density
D_{bw}	Wet weight bulk density
D_w	Density of water
d_w	Depth of water applied during surface flooding

d_{vz}	Depth of vadose zone wetted by application of a depth of water d_w
f_t	Adjustment factor in four-electrode equation
g	Acceleration of gravity
H	Total hydraulic head
h	Soil-water pressure (or suction)
h_{cr}	Critical pressure head of soil for wetting, used in Green-Ampt equation
Hw	Depth at water ponded above soil surface in Green-Ampt equation
J	Specific discharge; flux; Darcian flux
J_L	Flux at depth L in a soil
K	Hydraulic conductivity
$K(\theta)$	Hydraulic conductivity as a function of water content θ
K_d	Distribution coefficient
K_o	Steady-state hydraulic conductivity
L_f	Depth of wetting front in Green-Ampt equation
n	Porosity; also volume of mole of water
P_v	Volumetric water content of a soil
P_w	Moisture percentage of a soil on a dry weight basis
P/P_o	Relative humidity
Q	Total discharge in Darcy's equation
R_t	Measured resistance in ohms
r	Effective radius of soil pores
S	Sorptivity
S_t	Total porosity of a soil or vadose zone region
T	Surface tension of a liquid; absolute temperature; tortuosity
T_L	Time lag constant in piezometer equation
t	Time
V_i	Infiltration rate in Green-Ampt equation
v_i	Pore velocity of adsorbed species
V_s	Bulk volume of soil
V_w	Volume of water in a soil sample
v_w	Pore velocity of migrating water
$\bar{v}$	Average linear velocity; J/θ
W_d	Dry weight of soil
W_w	Wet weight of soil
x	Height of mercury column in tensiometer manometer
Y_o	Distance of sudden rise of water level in piezometer equation
y	Distance from soil surface to mercury reference level in tensiometer
z	Depth below land surface to the ceramic cup of a tensiometer
α	Contact angle between liquid and solid surfaces
ΔPw	Difference in percentage of water between two water contents
Δsm	Change in soil moisture storage
θ	Volumetric water content
Ψ	Soil-water potential
ρb	Particle density of solids

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SECTION 1

INTRODUCTION

Recent environmental legislation relative to the disposal of wastes has justifiably stressed the importance of protecting potable groundwater from waste-associated pollutants. Thus, the Safe Drinking Water Act of 1974 (PL 93-523) calls for the initiation of state programs to protect underground sources of drinking water. Similarly, the Federal Water Pollution Control Act of 1972 (PL 92-500) calls for the development of plans in designated areawide waste-management planning districts in order "... to control the disposal of pollutants on land or in subsurface water quality." The Resource Conservation and Recovery Act of 1976 (PL 94-580) recognized that "... open dumping is particularly harmful to health (and) contaminates drinking water from underground surface supplies ...."

The potential danger to potable water resulting from improper disposal of hazardous wastes was highlighted by the recent example of water pollution near the Love Canal in New York State. However, this case is but one of many in the United States, and the EPA has compiled over 400 case studies demonstrating the disastrous results of poor disposal of hazardous wastes (U.S. EPA, 1978).

In recognition of the need to safeguard groundwater resources, guidelines emanating from the principal environmental acts call for source and groundwater monitoring. In particular, the proposed guidelines relative to hazardous waste disposal (U.S. EPA, 1978) explicitly call for leachate and groundwater monitoring. Correspondingly, proposed guidelines for the landfill disposal of solid wastes (U.S. EPA, 1979b) specify that, "A groundwater monitoring system should be installed for the purpose of detecting the impact of all landfill disposal facilities which have the potential for discharge to an underground drinking water source." In addition to landfills, other waste disposal operations which may require monitoring include the following: pits, ponds, lagoons, septic tank areas, dry channels used for effluent disposal, dry wells, and injection wells. Monitoring irrigation return flow to groundwater is advisable in irrigated areas, particularly where heavy use of fertilizers and pesticides may cause pollution. Monitoring is an important consideration during disposal of spoils from coal strip-mining and oil shale developments (Everett, 1979; Slawson, 1979).

In a multiple-source area, a rating process may be necessary to rank sources in order of monitoring priority. Rating schemes have been proposed by LeGrand (1964); Pavoni, Hagerty, and Lee (1972); and Silka and Swearingen (1978). A systematic approach for rating potential sources and developing source-specific monitoring programs was proposed by Todd et al. (1976). This approach includes an evaluation of potential pollutants at each source location, considers the mobility of pollutants in the site-specific hydrologic regimen, and examines the potential impact of pollutants on current groundwater uses. Everett et al. (1976) presented a review of specific techniques and costs for establishing comprehensive monitoring networks. Similarly, Fenn et al. (1977) prepared a procedures manual to aid in designing monitoring systems at solid waste disposal facilities.

IMPORTANCE OF SOURCE MONITORING AND MONITORING IN THE VADOSE ZONE: WHEN SHOULD VADOSE ZONE MONITORING BE DISCOUNTED?

Figure 1 shows the hypothetical flow paths of a pollutant beneath a waste disposal site. It is well known that when groundwater underlying such a disposal site begins to manifest pollution by quality changes in pumped water, the aquifer may have already become badly contaminated. Monitoring the quality of wastes entering and within a disposal site (i.e., source monitoring), coupled with monitoring in the vadose zone, offers an "early warning" or groundwater pollution. Remedial measures may be implemented prior to the onset of severe pollution and the associated renovation costs will be eliminated or reduced. As an example, Wilson and Schmidt (1979) presented a case study entailing the monitoring of cascading water in an abandoned irrigation well near a field irrigated with sewage effluent. Groundwater from a nearby pumping well was also monitored. Analysis of cascading water samples showed that water of very poor quality was in transit through the vadose zone. Furthermore, the quality of cascading water was markedly poorer than that of pumped groundwater. The important feature of this case study is that if only pumped groundwater had been used to monitor return flow, it would have been erroneously concluded that no pollution was occurring.

In addition to providing an early warning of pollutant movement in groundwater, vadose zone monitoring may reduce the need for and costs of groundwater monitoring. In other words, if a vadose zone monitoring program fails to detect the movement of pollutants, the requirements for groundwater monitoring may be reduced or largely precluded. The savings in costs for constructing monitoring wells could be significant.

It is not our intention to proclaim vadose zone monitoring as a panacea for all hydrogeological conditions and for all waste disposal operations. The need for and extent of such monitoring should be tailored to site-specific conditions. For example, if the water table at a given site is relatively shallow, say within 10 feet of the land surface, vadose zone monitoring may be minimal. Similarly, if the vadose zone consists of fractured media, flow occurs primarily in channels and the ameliorating interactions of the vadose zone and water-borne pollutants may be minimal. Monitoring activities for these two cases could be restricted to source monitoring and groundwater monitoring. Installation of in-source monitoring facilities, including vadose zone units, may not be recommended for certain sources, such as landfills. The rationale was presented in the proposed guidelines for landfill disposal of solid waste (U.S. EPA, 1979b): "In no case should groundwater or leachate monitoring wells be installed through the bottom of the landfill proper since such installation could result in creation of a conduit for the direct passage

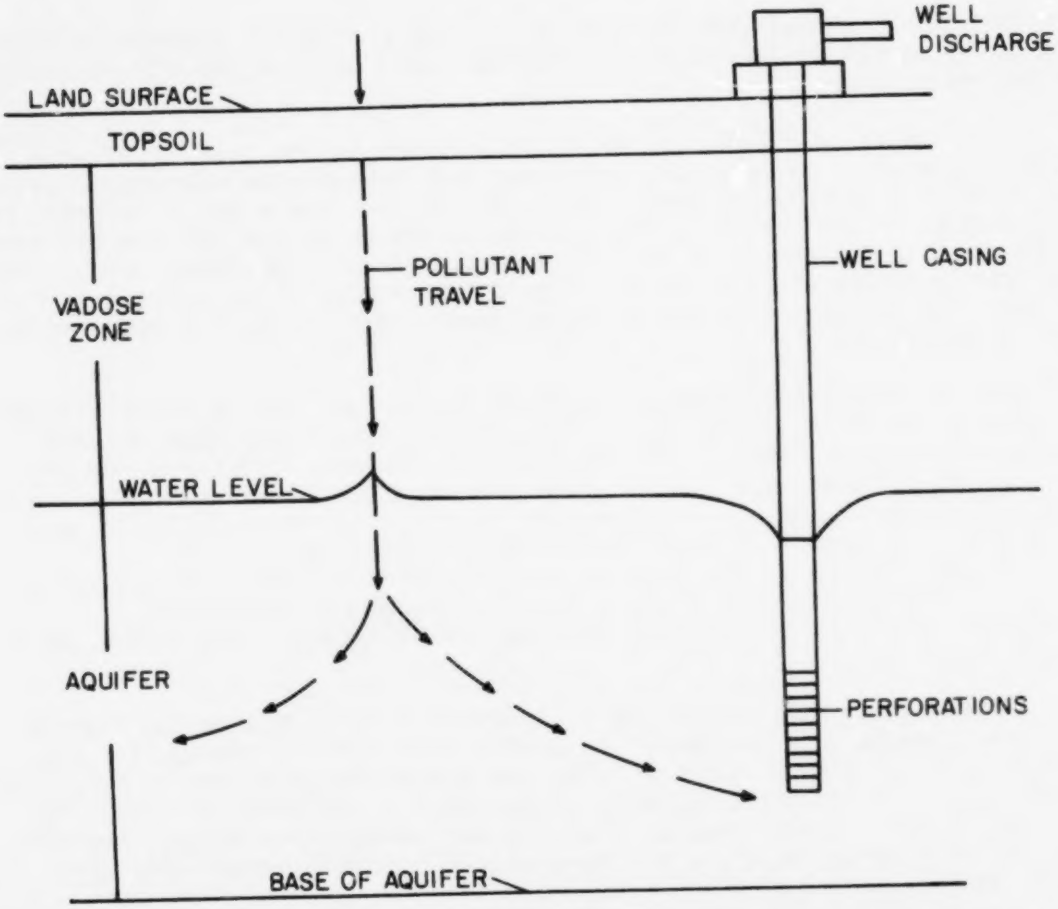
WELL
DISCHARGE
LAND SURFACE
TOPSOIL
POLLUTANT
TRAVEL
WELL CASING
VADOSE
ZONE
WATER LEVEL
AQUIFER
PERFORATIONS
BASE OF AQUIFER

Figure 1. Schematic representation of the flow paths of a pollutant from a surface source (after Schmidt, 1977).

of landfill leachate into underlying groundwater." Thus, it would be necessary to either install monitoring facilities around the periphery of the landfills or use angle-drilling methods to install such facilities below the base of the lowermost cells.

OBJECTIVE OF MONITORING IN THE VADOSE ZONE

The basic objective of a monitoring program in the vadose zone is to characterize three properties: (1) storage properties, (2) flow rates, and (3) spatial and temporal changes in pollutants. Storage properties are important in that pollutants are placed in temporary storage within the void space of the vadose zone prior to movement into groundwater. In some cases, such as in western valleys, the storage space may be sufficient to preclude movement into productive aquifers. In fact, Winograd (1974) suggested using the vadose zone for the storage of low-grade radioactive wastes. Mann (1976) recommended using the storage space of the vadose zone in arid regions to augment the above-ground storage of tailings ponds. Wilson, Osborne, and

Percious (1968) showed that by locating a disposal pond for blowdown effluent near an ephemeral stream, periodic recharge events diluted the effluent stored in the vadose zone.

Information on the flux of water (flux refers to the quantity of water flowing through a unit area per unit time) may indicate the macroscopic velocity of water in the vadose zone. Thus, if a piston flow model is assumed, the arrival time of pollutants at the water table may be determined from estimates of flux and the thickness of the vadose zone. (As will be shown later, a more realistic estimate of particle velocity is the average linear velocity, which accounts for variations in the cross-sectional areas of the flow path and degree of saturation.)

Pollutants may not necessarily travel at the same rate as water because of attenuation in the vadose zone. Consequently, the travel time of water represents an upper limit on the mobility of pollutants. (A few pollutants may move at a higher rate than water because of physical and chemical properties. For example, based on differences in viscosity, the velocity of gasoline should be about 1.5 times that of water (Schwille, 1967).) In considering rate, it is important to quantify both the infiltration rate at the ground surface and the percolation rate through the vadose zone. "Recharge" refers to the movement of water and pollutants across the water table.

Information on the third element, changes in pollutants during transit within the vadose zone, is needed to quantify the mobility and spatial distribution of specific pollutants. The vadose zone may attenuate certain pollutants to the point that they could be excluded as pollution sources. For example, Runnells (1976) demonstrated that soil from Sulfur Spring, New Mexico, has an enormous capacity to remove copper from mill water. The additional observation that copper removal was irreversible indicated that thousands of years would be required before groundwater (located at about 100 feet*) at the site would be affected.

PURPOSE OF THE REPORT

Present day monitoring in the vadose zone entails using a mix of methods developed by soil scientists and hydrogeologists. As a consequence, vadose zone monitoring requires an interdisciplinary approach. Unfortunately, many of those charged with designing, implementing, and regulating monitoring operations lack the necessary multidisciplinary skills and, in fact, may not even be aware that an interdisciplinary approach is required.

The purpose of this report is to provide an introduction to the principles and techniques of vadose zone monitoring. In the next section, a review of general features of the vadose zone is presented to set the stage for subsequent chapters. The third section reviews preliminary approaches to a vadose zone monitoring program. Subsequent sections review the technical elements and state of the art in present monitoring techniques in each of the three categories: storage properties in the vadose zone, flow properties in the vadose zone, and pollutant mobility in the vadose zone. The technical review in each of these sections is intended to provide the reader with a basic introduction to the theories governing the selection of specific methods and to review associated laboratory procedures. The reader who is not interested in such background information should pass directly to the review of field methods.

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*See Appendix A for conversion to metric units. English units are used in this report because of their currently common usage in industry and hydrology-related sciences. In cases where metric units are more appropriate (e.g., laboratory analyses), the English units are included parenthetically.

To facilitate comparison of the alternative techniques available for specific functions, advantages and disadvantages have been presented whenever possible. Regarding costs, the present rate of inflation precludes quoting precise values for the techniques described in this report. Instead, the reader is referred to a report by Everett et al. (1976) which reviews procedures for estimating the costs for many of the techniques under a range of hydrogeological conditions.

SECTION 2

GENERAL FEATURES OF VADOSE ZONES

The geological profile extending from ground surface to the upper surface of the principal water-bearing formation is called the vadose zone. As pointed out by Bouwer (1978), the term "vadose zone" is preferable to the often-used term "unsaturated zone" because saturated regions are frequently present in some vadose zones. The term "zone of aeration" is also often used as a synonym for vadose zone.

Davis and deWiest (1966) subdivided the vadose zone into three regions, designated as the soil zone, the intermediate vadose zone, and the capillary fringe. The surface soil zone is generally recognized as that region which manifests the effects of weathering of native geological material, together with the processes of elluviation and illuviation of colloidal materials, to develop more or less well-developed profiles (Simonson, 1957). The movement of water in the soil zone occurs mainly in the unsaturated state, i.e., that state in which the soil water exists under pressures less than atmospheric. The principal transport mechanisms associated with unsaturated flow are infiltration, percolation, redistribution, and evaporation (Klute, 1969). In some soils, primarily those containing horizons of low permeability, saturated regions may develop during surface flooding, creating shallow perched water tables.

The physics of unsaturated soil-water movement has been intensively studied by soil physicists, agricultural engineers, and microclimatologists. In fact, a copious amount of literature is available on the subject in periodicals (Journal of the Soil Science Society of America, Soil Science) and books (Childs, 1969; Kirkham and Powers, 1972; Hillel, 1971). Similarly, a number of published references are available on the theory of flow in perched water tables (Luthin, 1957; van Schilfgaarde, 1970). Soil chemists and soil microbiologists have also attempted to quantify chemical-microbiological transformations during soil-water movement (Bohn, McNeal, and O'Connor, 1979; Rhoades and Bernstein, 1971; Dunlap and McNabb, 1973).

Weathered materials of the soil zone may gradually merge with underlying deposits, which are generally unweathered, comprising the intermediate vadose zone. In some regions, this zone may be practically nonexistent, the soil zone merging directly with bed rock. In alluvial deposits of western valleys, however, this zone may be hundreds of feet thick. Figure 2 shows a geologic cross section through a vadose zone in an alluvial basin in California. By the nature of the processes by which such alluvium is laid down, this zone is unlikely to be uniform throughout, but may contain micro- or macrolenses of

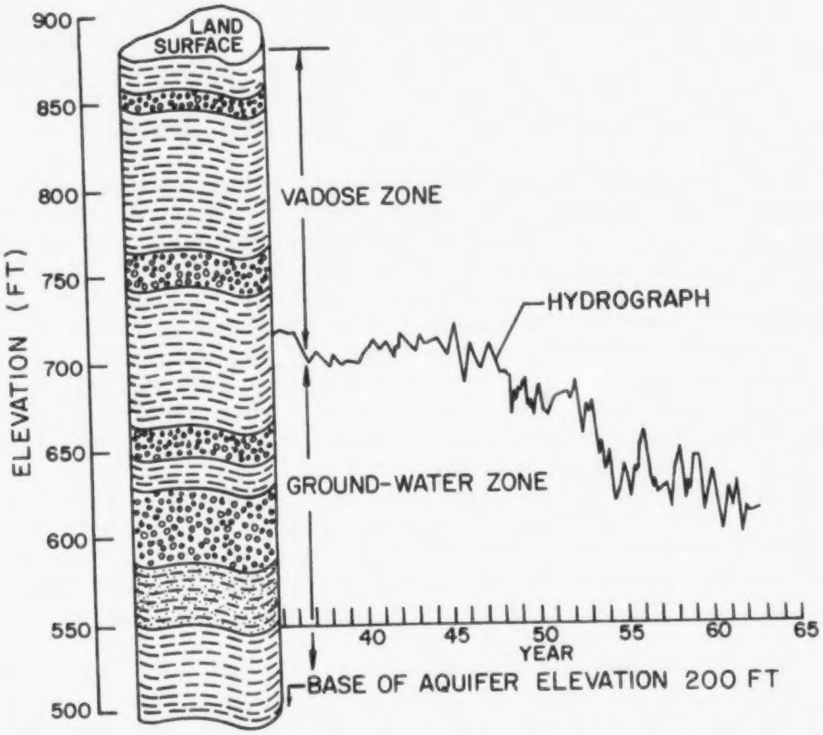


Figure 2. Cross section through the vadose zone and groundwater zone (after Ayers and Branson, 1973).

silts and clays interbedding with gravels and clays. Water in the intermediate vadose zone may exist primarily in the unsaturated state, and in those regions receiving little inflow from above, flow velocities may be negligible. However, perched groundwater may develop in the interfacial deposits of regions containing varying textures. Such perching layers may be hydraulically connected to ephemeral or perennial stream channels so that, respectively, temporary or permanent perched water tables may develop. Alternatively, saturated conditions may develop as a result of deep percolation of water from the soil zone during prolonged surface application of water. Studies by McWhorter and Brookman (1972) and Wilson (1971) have shown that perching layers that intercept downward-moving water may transmit the water laterally at substantial rates. Thus, these layers serve as underground spreading regions transmitting water laterally away from the overlying source area. Eventually, water leaks downward from these layers and may intercept a substantial area of the water table. Because of dilution and mixing below the water table, the effects of recharge may not be noticeable until a large volume of the aquifer has been affected.

In contrast to the large number of theoretical studies on water movement in the soil zone, parallel studies in the intermediate zone have been minimal. In fact, Meinzer (1942) coined the term "no-man's land of hydrology" to describe the limited knowledge of this zone. Reasoning from Darcy's equation, Hall (1955) developed a number of equations to characterize mound (perched groundwater) development in the intermediate zone. Hall also discussed the hydraulic energy relationships during lateral flow in perched groundwater. More recently, Freeze (1969) attempted to describe the continuum of flow between the soil surface and underlying saturated water bodies.

The base of the vadose zone, the capillary fringe, merges with underlying saturated deposits of the principal water-bearing formation. This zone is not characterized as much by the nature of geological materials as by the presence of water under conditions of saturation or near-saturation. Studies by Luthin and Day (1955) and Kraijenhoff van deLeur (1962) have shown that both the hydraulic conductivity and flux may remain high for some vertical distance in the capillary fringe, depending on the nature of the materials. In general, the thickness of the capillary fringe is greater in fine materials than in coarse deposits. Apparently, few studies have been conducted on flow and chemical transformations in this zone. Taylor and Luthin (1969) reported on a computer model to characterize transient flow in this zone and compared results with data from a sand tank model. Freeze and Cherry (1979) indicated that oil reaching the water table following leakage from a surface source flows in a lateral direction within the capillary fringe in close proximity to the water table. Because oil and water are immiscible, oil does not penetrate below the water table, although some dissolution may occur.

The overall thickness of the vadose zone will not necessarily be constant. For example, as a result of recharge at a water table during a waste disposal operation, a mound may develop throughout the capillary fringe extending into the intermediate zone. Such mounds have been observed during recharge studies (e.g., Wilson, 1971) and efforts have been made to quantify their growth and dissipation (Hantush, 1967; Bouwer, 1978). Pumping in the groundwater zone also modifies the overall thickness of the zone of aeration, as apparent on Figure 2.

As already indicated, the state of knowledge of water movement and chemical-microbiological transformations is greater in the soil zone than elsewhere in the vadose zone. Fortunately, renovation of an applied wastewater occurs primarily in the soil zone. This observation is borne out by the results of the well-known studies of McMichael and McKee (1966), Parizek et al. (1967), and Sopper and Kardos (1973). These studies indicate that the soil is essentially a "living filter" effectively reducing certain microbiological, physical, and chemical constituents to safe levels after passage through a relatively short distance (e.g., Miller, 1973; Thomas, 1973). As a result of such favorable observations, a certain complacency may have developed with respect to the need to monitor only in the soil zone. However, in one reported instance at a disposal operation for cannery wastes, an apparent breakdown of the living filter occurred because of excessive soil loading with the sugars, sucrose, and fructose (Meyer, personal communication, 1979). There is growing concern, moreover, that certain toxic substances, such as chlorinated hydrocarbons, other refractory organics, and microorganisms, may escape the soil zone, eventually passing into underlying groundwater (Walker, 1973; Ongerth et al., 1973; Shuval and Gruener, 1973; Robertson, Toussaint, and Jorque, 1974). For example, when soils are thin and underlain in the intermediate zone by fractured rocks, short circuiting of these materials may occur directly into groundwater. Thomas and Phillips (1979) have shown that for well-structured soils, rapid water movement occurs in macropores between soil peds. Consequently, entrained pollutants may not have the opportunity to interact with the bulk of the soil matrix.

Dunlap and McNabb (1973) also pointed out that microbial activity may be significant in the regions underlying the soil. They recommended that investigations be conducted to quantify the extent that such activity modifies the nature of pollutants travelling through the intermediate zone.

The foregoing discussion again emphasizes the need to monitor water movement and water quality throughout the vadose zone. For the soil zone, a large number of techniques were compiled by Black (1965). Monitoring in the intermediate zone and capillary fringe will require the extension of technology developed in both the soil zone and in the groundwater zone. Examples are already available where this approach has been used. For example, Apgar and Langmuir (1971) successfully used suction cups developed for in-situ sampling of the soil solution to sample at depths up to 50 feet below a sanitary landfill. Meyer, in a personal communication (1979), reported that suction cups were used to sample from depths greater than 100 feet below land surface at cannery and rock phosphate disposal sites in California.

In addition to the use of available techniques, new methods will undoubtedly be developed as additional insight is gained into the complexity of flow in the vadose zone.

SECTION 3

PRELIMINARY CHARACTERIZATION OF THE VADOSE ZONE

Prior to initiating a vadose zone monitoring program, an assessment should be made of existing monitoring efforts and relevant background information should be collected. Such a "nonsampling" approach will aid in the selection of methods to offset monitoring gaps and will generally increase the chances of designing an effective system. Assessing the existing monitoring includes determining sample collection methods for the source, defining the network of existing monitoring wells, including the locations and types of wells, and determining methods for sampling groundwater. Background information should be obtained on the following items: (1) nature of pollutants, (2) pollutant source loading, (3) existing groundwater usage and quality, and (4) a description of the hydrogeologic framework, including infiltration potential and hydraulic/geochemical properties of the vadose zone and groundwater zone.

The existing source monitoring program could be evaluated by comparing the ongoing sampling techniques with recommended approaches described in recent published guidelines, such as the report of Huibregtse and Moser (1976). Similarly, the groundwater monitoring program could be compared with approaches recommended by Mooij and Rovers (1976), and by Everett et al. (1976). Approaches for collecting background quality information on a source and existing groundwater usage and quality have been described in detail by Todd et al. (1976) and by Fenn et al. (1977). The quality data should be examined for completeness; for example, there may be gaps in existing analyses. Specific pollutants should be delineated and ranked.

As a first step in characterizing the hydrogeological framework with particular emphasis on the vadose zone, all relevant data should be collected. For example, comprehensive soil survey data may be available through state agricultural experiment stations, the Soil Conservation Service, or other agencies. Such surveys generally include information on depths of soil horizons, presence of hardpan, clay pan, color, porosity, structure, texture, organic matter content, soil pH, and infiltration rates.

For deeper horizons in the lower vadose zone, data may be available from drillers' well logs, Water Supply Papers of the U.S. Geological Survey, University theses, dissertations, and special reports. In general, well logs by drilling companies mainly describe conditions below the water table. However, they should be consulted for possible clues on stratification. In some areas, the U.S. Geological Survey collects drill cuttings from drilling sites. These samples are evaluated in the laboratory for particle-size distribution, color, and source of origin. Some universities also collect samples as part of their hydrology or agricultural engineering programs. Again, the purpose of these sampling programs is to characterize the saturated zone, but valuable clues on the vadose zone may also be included (Matlock, Morin, and Posedly, 1976). Historic trends in groundwater levels should be determined by examining existing water level contour maps.

In addition to collecting available data, wells in the area of a monitoring site should be used to the fullest extent. For example, static water levels in a network of wells can be plotted to give the water table configuration and thus an approximate idea of the thickness of the vadose zone. Also, the presence of cascading water in wells will indicate the possible presence of perching layers. Finally, abandoned wells, or wells in which the pumps have been removed for servicing, may be used with borehole geophysical methods (described later) to provide further clues on the vadose zone.

After exhausting all existing sources for clues on the nature of the vadose zone at a site, a test drilling program may be considered necessary to provide more detailed information. The extent and thoroughness of the program will depend on the availability of funds. At any rate, care should be exercised in setting up the program to ensure the maximum value of the data. For example, a careful grid should be laid out for the project area. Also, the test wells should be established based on the possibility that they may eventually be used as observation wells, piezometers, or access wells during an actual monitoring program.

In general, samples from the soil zone are obtained to: (1) characterize the average soil texture, water content, or chemistry in increments (say 6-inch increments); (2) observe the precise depth-distribution of soil texture; or (3) determine the bulk density or water-release curves of soil increments. For the first purpose, samples obtained using post-hole augers, screw or sleeve-type augers, or power-driven augers are useful. For the second purpose, cores are obtained by driving small-diameter tubes into the soil to the desired depth. For the third purpose, larger-diameter core samplers are used. These may be hand-driven or forced into the soil by power-driven hydraulic soil coring equipment. With these methods, cores of a specific volume are obtained. More information on coring is presented by Blake (1965).

To sample throughout the vadose zone, it may be necessary to construct deep wells, using standard drilling techniques. Such techniques include jetting, rotary, cable-tool, augering, and air drilling. Of these methods, perhaps augering, using continuous flights, and air drilling provide the most usable samples. Problems develop in characterizing the distribution of indigenous salt and water content with cable-tool and rotary methods because of water additions during the drilling process.

One air drilling technique involves driving a double-wall drive tube by a pile hammer and, concurrently, forcing air under pressure down the annulus of the pipe. Air and entrained material cut by the bit return to the surface through the inside pipe. The sample, available continuously, is diverted into a cyclone sampler where it is bagged for laboratory analyses. According to the manufacturer, formation changes can be determined within a few centimeters. Furthermore, water seams can be determined immediately. This feature is advantageous in locating the depth and thickness of perching layers. Whichever technique is used, samples should be taken in increments throughout the vadose zone.

Drill cuttings could be examined in the laboratory for particle-size distribution and other geologic parameters. Knowing the vertical distribution of grain size in the hole, the location of potential perching zones may be delineated. The collected samples could also be used to estimate storage and transmissive properties of the vadose zone.

Following construction of a test well or wells, geophysical logging should be initiated. First, the test hole(s) should be logged. The results of logging in the test well should be correlated with drill cutting analyses to delineate stratigraphy. Subsequent logs from other wells could thus be interpreted and the lateral and vertical extent of various layers (e.g., those favoring perching) defined.

Among the common borehole logging techniques are the spontaneous potential method, resistance logging, acoustic logging, and nuclear logging. The spontaneous potential method and resistance logging require an uncased water-filled well. Acoustic logging is conducted in a fluid-filled cavity and is not applicable to the vadose zone. Only nuclear logging methods are suitable for logging in either uncased or cased wells above the water table.

Common nuclear logging techniques include natural gamma logging, gamma-gamma logging, and neutron logging. The principles of and procedures in applying these techniques in groundwater investigations are discussed thoroughly by Keys and MacCary (1971). Briefly, natural gamma logging comprises the detection of natural gamma activity of rocks. Keys and MacCary (1971) list the following sequence of sedimentary rocks in ascending order of natural gamma activity: anhydrite, coal, salt, dolomite, sandstone, sandy shale, shale, organic marine shale, and potash beds. Natural gamma logs are particularly advantageous in characterizing the vertical extent of sediments and tracing them laterally from well to well (Keys and MacCary, 1971; Norris, 1972).

Gamma-gamma logs provide a record of the intensity of radiation from a source in a down-hole probe after it has backscattered and attenuated in the well and surrounding media (Keys and MacCary, 1971). The down-hole probe contains a source of gamma photons, such as cobalt-60 or cesium-137, and sodium iodide detector. The principal uses of gamma-gamma logs are to identify the lithology and to estimate bulk density and porosity of rocks (Keys and MacCary, 1971).

Neutron logging comprises lowering into a well a down-hole probe containing a source of high-energy neutrons and a detector of thermalized neutrons. Neutron loggers provide information on the hydrogen content, and consequently water content, of sediments within the vadose zone and the porosity of sediments below the water table. Neutron logs are useful in detecting the presence of perched groundwater. For example, Keys (1967) used a combination of gamma logs and neutron logs to delineate a clay perching bed at the National

Reactor Testing Station in Idaho. Neutron logging between wells may indicate the lateral extent of perched groundwater bodies.

SECTION 4

MONITORING STORAGE PROPERTIES OF THE VADOSE ZONE

TECHNICAL CONSIDERATIONS

Physical properties of the vadose zone associated with storage of water include the following: (1) total thickness, (2) porosity, (3) bulk density, (4) water content, (5) soil-water characteristics, (6) field capacity (specific retention), (7) specific yield, and (8) fillable porosity. Greater technical information on each of these items may be found in reference works by Childs (1969); Hillel (1971); Davis and deWiest (1966); Cooley, Harsh, and Lewis (1972); Freeze and Cherry (1979); Bouwer (1978); and Brakensiek, Osborn, and Rawls (1979).

The relationships of each of these properties to storage are reviewed in this section. Laboratory methods for quantifying the magnitudes of the first six properties are also presented.

Thickness

The storage capacity of a vadose zone is obviously related to the overall thickness, i.e., depth from ground surface to the water table. As shown on Figure 2 , the thickness is not constant but changes in response to pumping or recharge. The theoretical depth of water that could be stored in a vadose zone assuming 100-percent saturation is expressed by the following relationship:

$$d_w = \frac{\Delta P_w}{100} \frac{D_b}{D_w} d_{vz} \tag{1}$$

where d_w = depth of water applied during surface flooding

D_b = bulk density of soil

D_w = density of water

d_{vz} = depth of vadose zone to be wetted

ΔP_w = difference in the percentage of water on a mass basis at saturation and after oven drying (105°C for 24 hours).

As an example of the use of this equation to estimate the theoretical maximum storage capacity of a vadose zone, consider the following hypothetical conditions:

1. 100-foot thick vadose zone, uniform and unlayered, i.e., d_{vz} = 100 feet
2. Bulk density of media = 81.1 lb/ft^3 = 1.3 gm/cm^3
3. Density of water = 62.4 lb/ft^3 = 1.0 gm/cm^3
4. Water content of drained vadose zone material = 10 percent (dry weight basis)
5. Water content of vadose zone material assuming 100-percent saturation = 30 percent (dry weight basis).

Using equation 1, the depth of water, d_w, that could be stored in the 100-foot thick hypothetical profile would be:

$$d_w = \frac{30 - 10}{100} \times 1.2 \times 100$$

$$= 26 \text{ feet.}$$

In other words, 26 acre-feet of water per acre of surface area would theoretically be required to saturate the 100-foot vadose zone. In reality, the vadose zone would never attain 100-percent saturation, even for an ideal media, because of entrapped air and because of drainage of the media to field capacity (see below).

In a multilayered media, the individual bulk densities of the various layers would need to be considered in using equation 1 to calculate total storage. Note also that only vertical storage is considered by this relationship. For point and line sources, storage should also be accounted for in a lateral direction.

Bulk Density

As shown by equation 1, bulk density values are required in order to estimate total water storage in the vadose zone. Bulk density represents the density of a soil or rock, including solids plus voids, after drying (Bouwer, 1978). To determine the bulk density of a material, field cores of a precise volume are oven-dried at 105°C until constant weight. The bulk density is the oven-dry mass divided by the sample volume. Blake (1965) and Paetzold (1979) also reviewed the alternative "clod method" for determining bulk density of disturbed samples.

Bulk density values range from 1.1 gm/cm^3 (68.6 lb/ft^3) for clay soils to 1.6 gm/cm^3 (100 lb/ft^3) in sandy soils (Hillel, 1971).

Porosity

As defined by Bouwer (1978), the porosity of a soil or rock material is the percentage of the total volume of the medium occupied by pores or other openings. Thus, the total porosity of a soil is a measure of the amount of water which could be stored under saturation. In actuality, because of entrapped air, saturation may not be reached until after a long period of time.

According to Vomicil (1965), total porosity may be calculated by the following expression:

$$S_t = 100\,[1 - (D_b/\rho_b)] \tag{2}$$

where S_t = total porosity, the percentage of the bulk volume not occupied by solids

D_b = bulk density

ρ_b = particle density, approximately 2.65 gm/cm^3 (165.4 lb/ft^3) for mineral soil.

Typical values of porosity for representative earth materials are presented in Table 1 and on Figure 3.

Laboratory estimation of the total porosity of a soil core requires determining the bulk density and particle density. Methods for determining the bulk density were reviewed above. Particle density is measured precisely using the pycnometer method (Vomicil, 1965).

Although a knowledge of the total porosity is of value when considering storage in the vadose zone, also of importance is the pore-size distribution. For example, sandy soils have primarily large pores of nearly equal size permitting easy drainage (U.S. EPA et al., 1977). However, medium-textured loamy soils have a greater porosity than sands and also a wider pore-size distribution. "Thus, more water is held at saturation in soils than in sands and it is removed much more gradually as matric potential becomes greater" (U.S. EPA et al., 1977).

The effect of macropores on retention and drainage of water in soils is receiving renewed attention (Thomas and Phillips, 1979). The effect of macropores is particularly important in well-structured soils. Water movement occurs rapidly in the interstices between soil peds and little storage may occur within the pores of the soil blocks.

Pore-size distribution is commonly found by means of water characteristic curves (discussed below).

Water Content

The total porosity of vadose zone materials represents the upper limit for water storage in the vadose zone. Unless the vadose zone has not been wetted since deposition, a certain residual amount of water remains in the

TABLE 1. RANGE OF POROSITY VALUES IN UNCONSOLIDATED DEPOSITS ON ROCKS (after Freeze and Cherry, 1979)

	Porosity (percent)
Unconsolidated Deposits	
Gravel	25 to 40
Sand	25 to 50
Silt	35 to 50
Clay	40 to 70
Rocks	
Fractured basalt	5 to 50
Karst limestone	5 to 50
Sandstone	5 to 30
Limestone, dolomite	0 to 20
Shale	0 to 10
Fractured crystalline rock	0 to 10
Dense crystalline rock	0 to 5

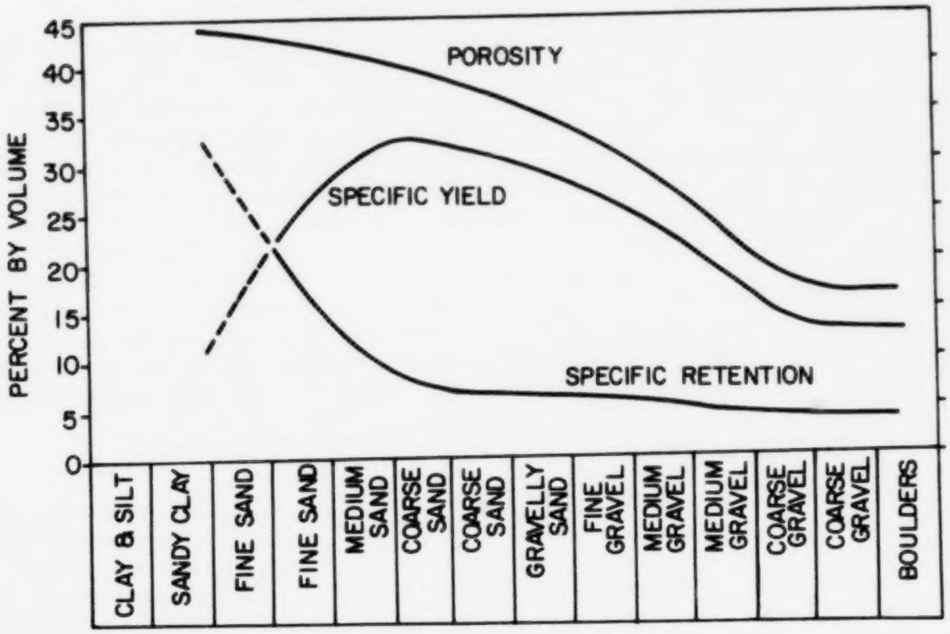


Figure 3. Variation of porosity, specific yield, and specific retention with grain size (after Scott and Scalmanini, 1978).

pores. Thus, the available storage is diminished by this residual water content. Water content is expressed on either a mass or a volume basis using the following expressions presented by Brakensiek, Osborn, and Rawls (1979):

$$P_w = \frac{W_w - W_d}{W_d} \times 100 \qquad (3)$$

and

$$P_v = \frac{V_w}{V_s} \times 100 \qquad (4)$$

where P_w = moisture percent on a dry weight basis

W_w = wet weight of soil

W_d = dry weight of soil

P_v = volumetric moisture percentage

V_w = volume of water in sample

V_s = bulk volume of soil.

As pointed out by Bouwer (1978), the volumetric water content is more useful for hydrological purposes than is the mass water content. When expressed as a fraction, the volumetric water content is denoted by the symbol θ. The basic laboratory technique for determining water content of a soil consists of (1) weighing the sample, (2) drying the sample for a set period of time (e.g., at 105°C for 24 hours), and (3) reweighing the dried sample. The above relationships are then used to calculate P_w and P_v. An alternative method entails placing a soil sample in a sealed container with calcium carbide reagent. The calcium carbide reacts with the water in the sample producing a gas. The gas pressure, registered on a gauge, is conducted into water content on a wet weight basis. Combined drying and weighing units are also commercially available for laboratory determination of water content. Gardner (1965) reviewed other methods, including the alcohol drying technique.

Soil-Water Characteristics

The theoretical saturation value of a soil is the total porosity (discussed above). As a soil drains, a range of soil-water content values is possible depending on various forces acting on the solids-water system. According to Day, Bolt, and Anderson (1967), these forces include matric forces, osmotic forces, and body forces (e.g., gravity). Matric forces include forces related to the surface tension of water, the cohesion of water molecules, the adhesion of water molecules to the surface of soil particles, and electrical forces at the molecular level (U.S. EPA et al., 1977). When expressed on a unit volume basis, the energy associated with these forces is termed soil-water pressure. The most common method for expressing the energy is on a unit weight basis, and the energy is termed soil-water pressure head, or head (U.S. EPA et al., 1977). The relationship between soil-water pressure and water content is called the soil-water characteristic. According to Klute (1969), the change in water content per unit pressure head change ($d\theta/dh$) is called the water capacity.

As pointed out by the U.S. EPA et al. (1977), "The force by which water is held in the soil is approximately inversely proportional to the pore diameter. Thus, the larger the pore, the less energy is required to remove water. As soil dries or drains, water is removed from the larger pores first. The water remains in the smaller pores because it is held more tightly. Thus, as soil-water content decreases, soil-water potential increases." Hillel (1971) pointed out that the amount of water retained at low negative head values depends on capillary effects and pore-size distribution and, consequently, is strongly affected by soil structure. In contrast, at higher netative heads, water retention is influenced less by structure and more by texture and specific surface.

The effect of soil texture on the shape of the soil-water characteristic curve is shown on Figure 4 where curves are presented for a clay soil and for a sandy soil. The curve for the clay soil is more gradual and the water content at a given negative head is greater than for the sandy soil. In the sandy soil, however, most of the pores are large and, once drained, the amount of water retained is small (Hillel, 1971).

To obtain soil-water characteristic curves in the laboratory down to negative pressures of -800 cm (315 inches) of water, the modified Haines method may be used for individual core samples. This method employs equipment shown on Figure 5. Soil cores are carefully placed on the fritted glassbead plate and saturated. After ensuring that the samples are saturated, excess water is removed and the hanging water column is adjusted through a desired range of negative heads. The cores are weighed after the soil water has equilibrated with each successive negative head. At the end of the test, the soil cores are oven-dried and intermediate masses are converted to volumetric water content values.

A plot of water content versus applied negative head for a desorption cycle (obtained via the Haines method) is different from the curve obtained during a wetting cycle. The difference between wetting and drying curves for a sand is shown on Figure 6. Note that the water content values at a given negative head are generally less during the wetting cycle than those during the drying cycle. Such hysteresis is caused by entrapment of air in the pore space during wetting. The figure also exhibits "scanning curves" which represent the change in the water content-negative pressure head relationship if the soil is suddenly wetted during drainage, or vice-versa.

Topp and Zebchuck (1979) presented a variation of the hanging column method which permits the simultaneous measurement of soil-water desorption curves from a number of large-diameter cores. The soil cores contact a "tension medium" in a tank. The medium comprises glass beads with a narrow pore-size distribution, permitting high hydraulic conductivity and high air-entry values. Negative heads up to -100 cm (-40 inches) of water are applied to the

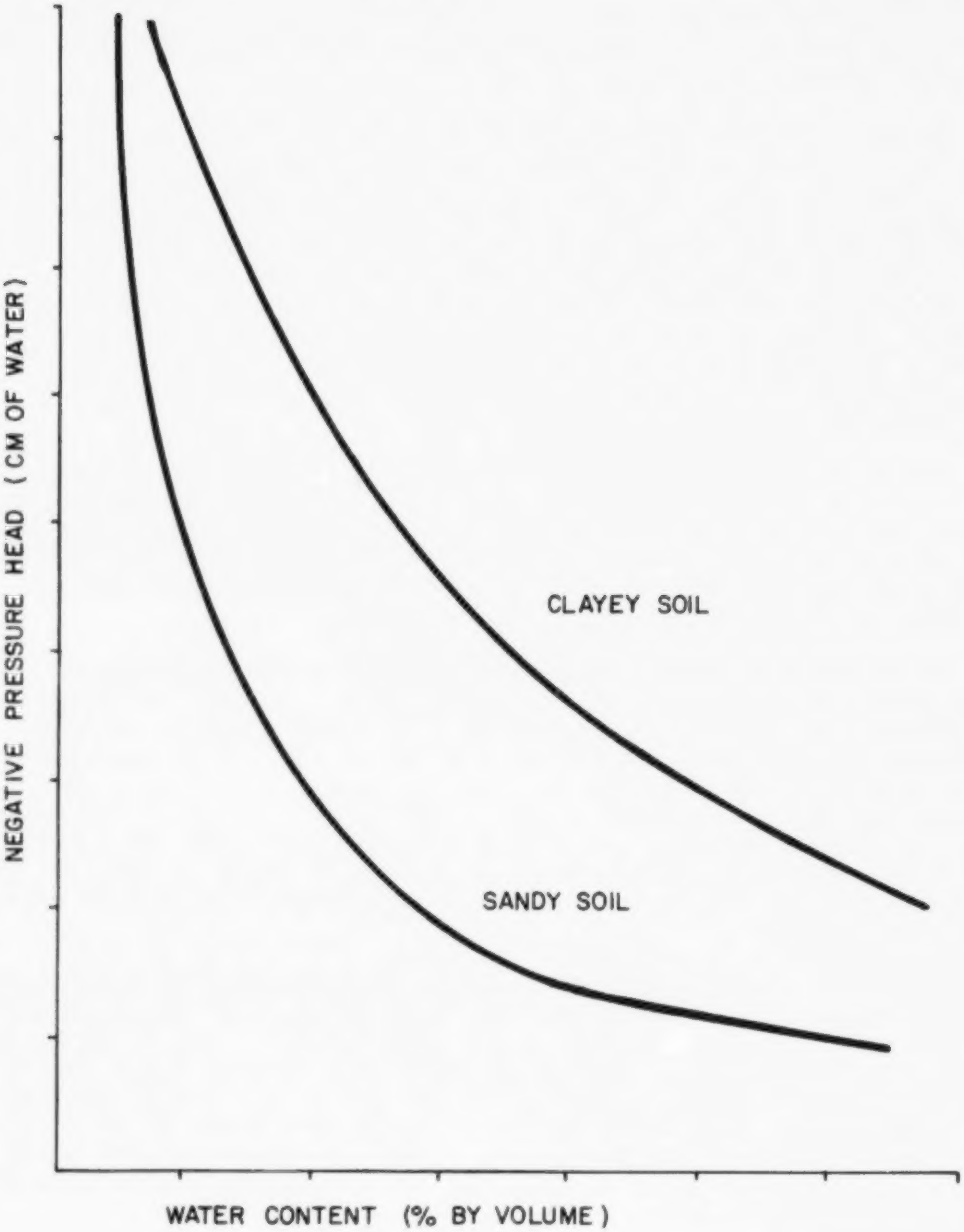


Figure 4. Soil-water characteristic curve for a clayey soil and sandy soil (after Hillel, 1971).

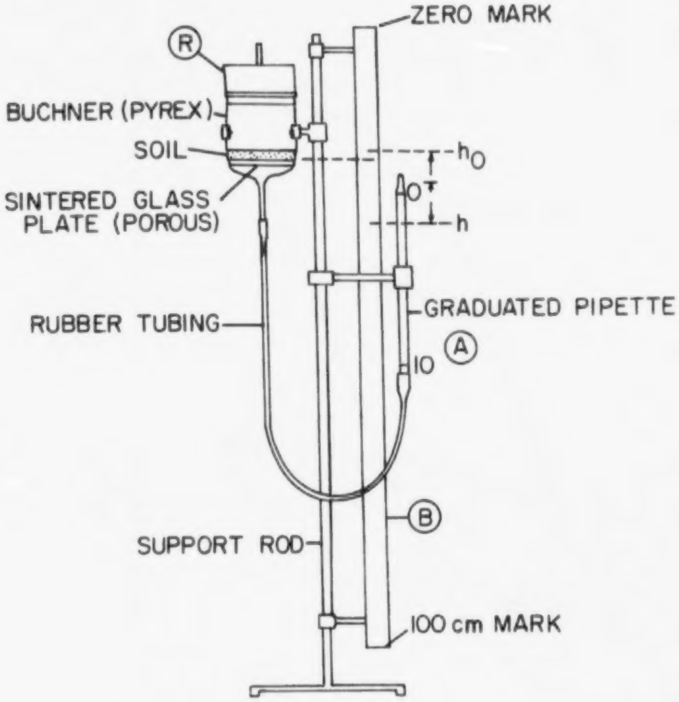

ZERO MARK
Ⓡ
BUCHNER (PYREX)
SOIL
SINTERED GLASS PLATE (POROUS)
h_0
10
h
GRADUATED PIPETTE
RUBBER TUBING
10
Ⓐ
Ⓑ
SUPPORT ROD
100 cm MARK

Figure 5. Modified Haines apparatus for obtaining soil-water characteristic curves (after Day, Bolt, and Anderson, 1967).

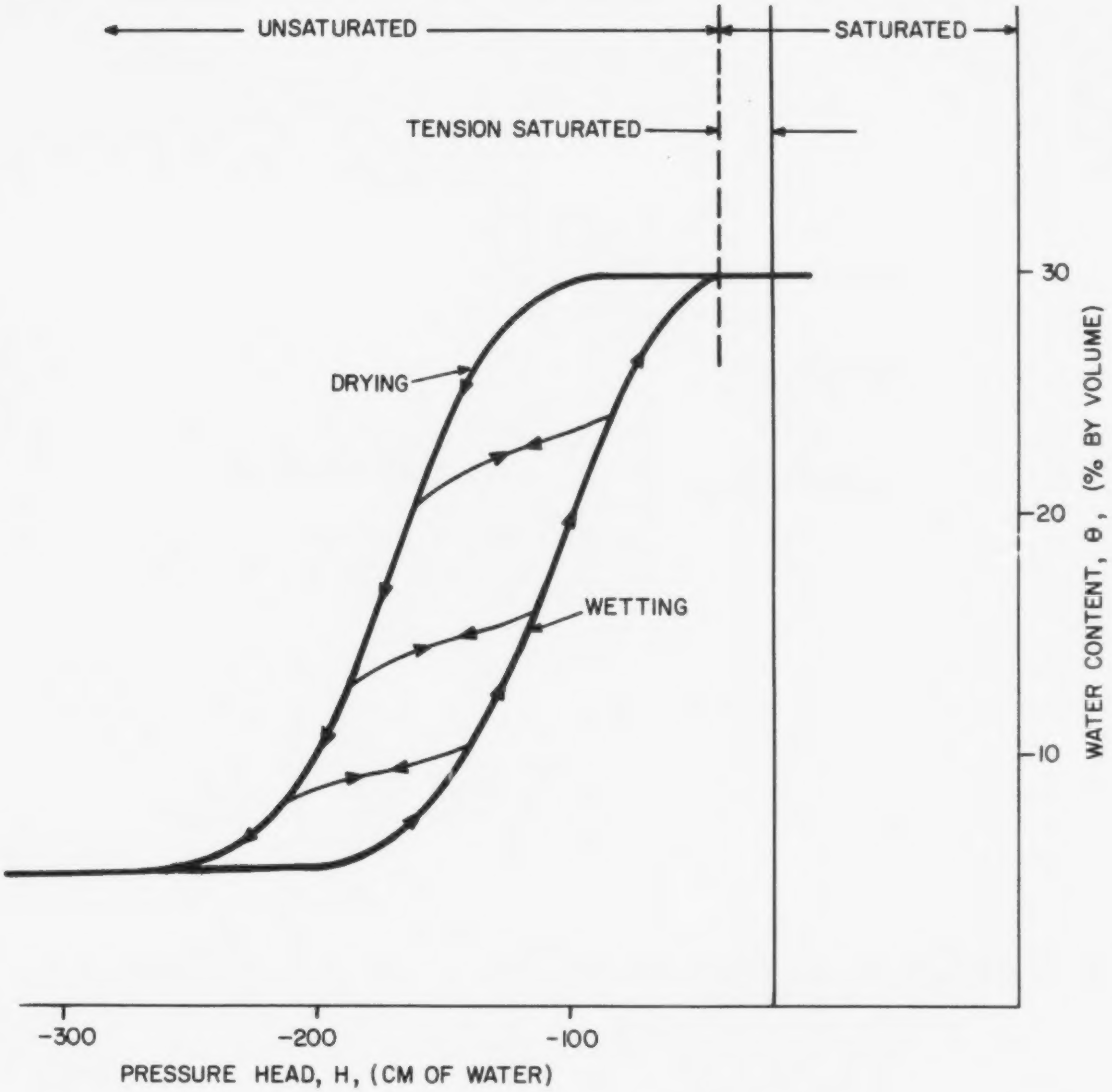


Figure 6. Soil-water characteristic curves for a sandy soil during wetting and drying showing the effects of hysteresis (after Freeze and Cherry, 1979).

tension medium via a hanging water column. Heads from -100 cm (-40 inches) to -500 cm (-200 inches) are obtained using a regulated vacuum. After equilibrating at successive pressures, the cores are weighed. At the end of the extraction period, the cores are oven-dried and intermediate masses are converted to volumetric water content values.

Inasmuch as the air-entry value of the tension medium for the above methods is about 0.8 atmosphere of negative pressure, other methods are required to obtain water content versus head relationships for greater negative head values. Commonly, the pressure-plate method is employed. Details of the method are presented by Richards (1965). As described by Paetzold (1979), the soil sample is wetted and placed in a pressure chamber on a ceramic plate or a cellulose-acetate membrane. A positive air pressure is applied to the soil within the chamber causing water to flow from the sample through the membrane. The air pressure value corresponds to the negative pressure head retaining water in the sample. The various air-entry values of membranes permit a range of possible pressures (for example, up to 15 atmospheres for ceramic plates). After the water content of the sample equilibrates at the applied air pressure, the sample is removed and weighed.

Soil-water characteristic curves may be used to construct the associated pore-size distribution--water content curves. The relationship between suction, h, and effective pore-size diameter is approximated by the following equation:

$$h = 2T \cos \alpha / \rho g r \tag{5}$$

where T = surface tension of the liquid

α = contact angle

ρ = density of liquid

g = acceleration of gravity

r = effective radius.

Field Capacity (Specific Retention)

Field capacity may be defined in a general sense as the volume of water which a unit volume of soil will retain against the force of gravity during drainage. The concept of field capacity was developed many years ago by agriculturists concerned with quantifying the amount of water to apply to irrigated fields. The original premise was that field capacity is a fixed value representing the amount of water stored in a soil a certain time after drainage has "essentially ceased." By the same token, it is usually assumed that during recharge (wetting), water movement will not occur until the medium has been wetted to field capacity. Although these concepts of field capacity are useful in an applied sense, one should be aware of certain technical limitations. Hillel (1971) discussed such limitations in detail. Briefly, one limitation is that the simplistic concept of field capacity fails to account for the dynamic nature of soil-water movement. In particular, drainage does not really cease at field capacity but may continue at a slower rate for a prolonged period of time. That is, "The redistribution process is in fact continuous and exhibits no abrupt 'breaks' or static levels. Although its rate decreases constantly, in the absence of a water table the process continues and equilibrium is approached, if at all, only after very long periods" (Hillel, 1971). The modern conception of field capacity is that it is not a unique soil property; instead, a range of values is possible.

For sandy soils, field capacity may be reached in a few hours. For soils finer than sandy soils (e.g., sandy loams), 2 or 3 days may be required to reach field capacity, and for medium- to fine-textured soils, a week may be required. For poorly structured clays, the time will be much greater (U.S. EPA et al., 1977). Approximate values of field capacity (on a mass basis) vary from 4 percent in sands to 45 percent in heavy clays and up to 100 percent or more in organic soils (Hillel, 1971). In terms of matric potentials, field capacity values for sands range from 0.1 to 0.15 bar (1 bar = 0.9869 atmosphere). For medium- to fine-textured soils, the corresponding range is 0.3 to 0.5 bar (U.S. EPA et al., 1977). The value of 0.3 bar is chosen as an average value.

Knowing the water content values of given soil at field capacity and the observed water content value at a given time, the depth of water applied at the land surface to bring the soil to field capacity may be calculated from equation 1. If the soil is layered, it will be necessary to account for the sum of the water contents of individual layers (see Brakensiek, Osborn, and Rawls, 1979, pp 523 and 524).

Among the factors affecting the apparent field capacity are the following (Hillel, 1971): (1) soil texture, (2) type of clay (e.g., clays predominantly comprised of the montmorillonite type exhibit a higher water holding capacity at field capacity), (3) organic matter content (the higher the organic matter level, the higher will be the field capacity), (4) antecedent water content, (5) presence of impeding layers, and (6) evapotranspiration. Soil structure is also an important factor in evaluating field capacity inasmuch as the large interpedal cracks permit more rapid drainage than the micropores within the soil blocks.

The water content of a soil sample at 0.3 bar is obtained in the laboratory using the pressure membrane method. This method was discussed above in the section on water characteristics. An alternative method to estimate field capacity is to assume that field capacity equals one half of the percent of water content at saturation. That is, $F_C = SP/2$. Saturation percentage is measured in the laboratory by determining the number of grams of water to saturate 100 grams of air dry soil (U.S. EPA et al., 1977).

The above discussion relates to the concept of field capacity as employed by agriculturists. The parallel term used by hydrogeologists is "specific retention," defined as the "quantity of water per unit total volume which will not drain under the influence of gravity" (Cooley, Harsh, and Lewis, 1972). Specific retention may be visualized as the water remaining in the dewatered region of the vadose zone after recession of the water table (Figure 7).

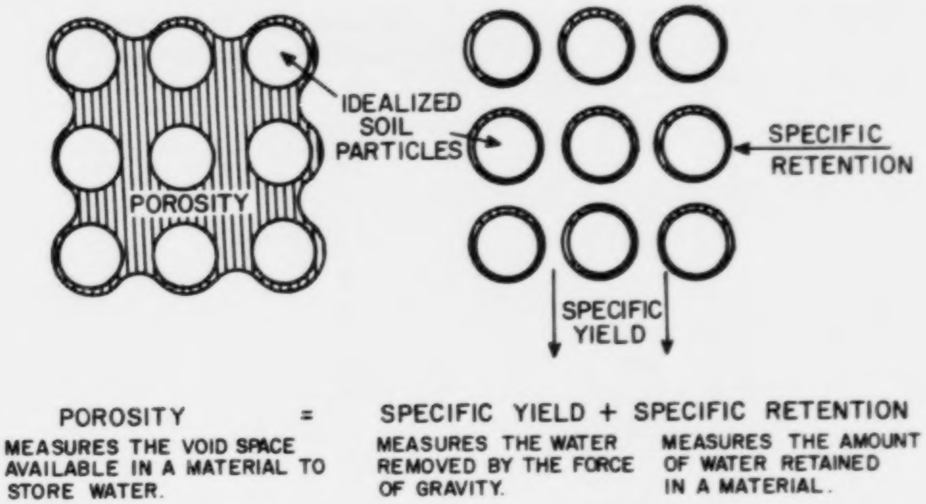

IDEALIZED SOIL PARTICLES
SPECIFIC RETENTION
POROSITY
SPECIFIC YIELD
POROSITY = SPECIFIC YIELD + SPECIFIC RETENTION
MEASURES THE VOID SPACE AVAILABLE IN A MATERIAL TO STORE WATER.
MEASURES THE WATER REMOVED BY THE FORCE OF GRAVITY.
MEASURES THE AMOUNT OF WATER RETAINED IN A MATERIAL.

Figure 7. Schematic representation of porosity, specific yield, and specific retention (after Scott and Scalmanini, 1978).

Specific Yield

"Specific yield" is a term employed by hydrogeologists to characterize storage in an unconfined aquifer. That is, specific yield is "... the volume of water that an unconfined aquifer releases from storage per unit surface area of aquifer per unit decline in the water table" (Freeze and Cherry, 1979). Figure 7 shows the conceptual relationship between specific yield and specific retention. As shown on Figures 3 and 7, the specific yield for a media equals the porosity value minus the value of specific retention. Representative specific yield values for valley sediments in California are shown on Table 2.

Fillable Porosity

The volume of water that an unconfined aquifer stores during a unit rise in water table per unit surface area is called the fillable porosity (Bouwer, 1978). As shown on Figure 8, the amount of water placed into storage (fillable porosity) during the rise of a water table is less than the corresponding amount released during drainage (specific yield). The difference in storage reflects hysteresis caused by air entrapped in pore sequences during the rise of the water table.

FIELD METHODS FOR DETERMINING STORAGE PROPERTIES OF THE VADOSE ZONE

As pointed out in a previous section, field methods for determining the storage properties of the vadose zone are a mixture of techniques developed by soil scientists and hydrogeologists. Field methods are reviewed for measuring the following characteristics: (1) thickness, (2) porosity and bulk density, (3) water content, (4) soil-water characteristics, (5) specific retention, and (6) specific yield and fillable porosity.

TABLE 2. COMPILATION OF SPECIFIC YIELD VALUES FOR VARIOUS MATERIALS IN CALIFORNIA VALLEYS (after Cooley, Harsh, and Lewis, 1972)

Material	Average specific yield (percent)
Clay	2
Silt	8
Sandy clay	7
Fine sand	21
Medium sand	26
Coarse sand	27
Gravelly sand	25
Fine gravel	25
Medium gravel	23
Coarse gravel	22

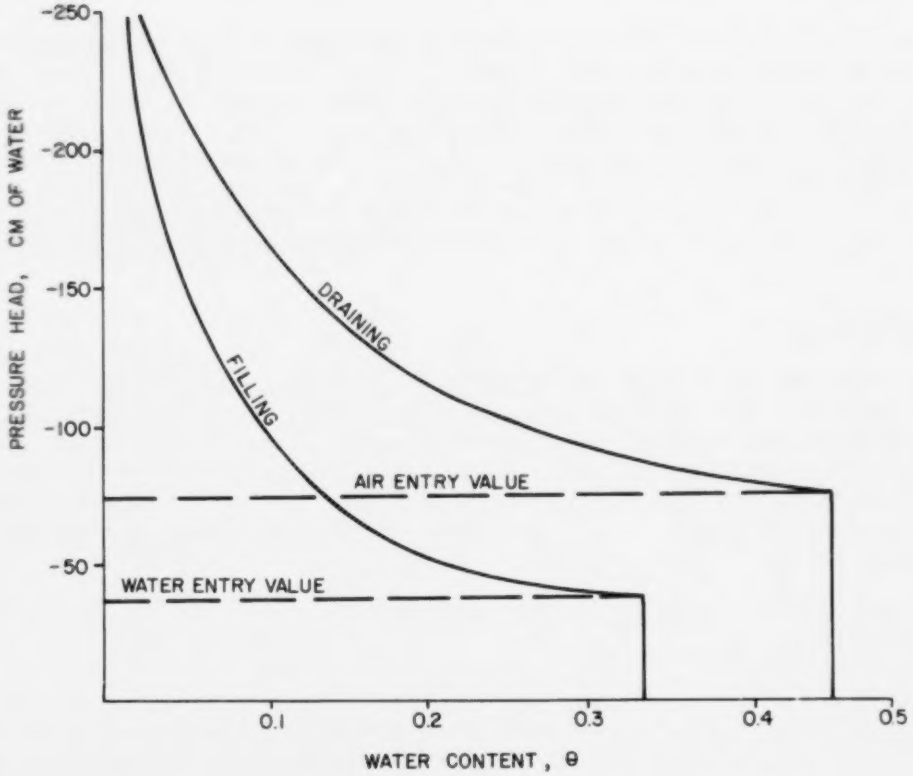


Figure 8. Water content distribution above a water table during a fall (draining) and rise (filling) in the water surface.

Thickness

The thickness of the vadose zone underlying the pollution source may be estimated directly by determining water levels in the area using either existing wells or specifically constructed wells. If the area is underlain by perched groundwater which is a source of potable water, the depth to the perched water table should be selected as the depth of the vadose zone (Silka and Swearingen, 1978). For confined leaky aquifers, the depth to the overlying unconfined water table should be selected. For nonleaky confined aquifers, the distance to the base of the uppermost confining layer could be used. When estimating storage for pollutants, it is advisable to use the minimum depth to groundwater, determined by examining historic water levels.

Indirect methods have been employed to detect the presence of shallow perched groundwater, thus providing an estimation of the depth of the vadose zone. For example, Estes and Simonett (1978) reported on using three imagery techniques to detect perched groundwater: visible and reflected infrared imagery, thermal infrared imagery, and microwave techniques. They evaluated the potential of these techniques for detecting perched water tables within 5 to 10 feet of the surface in the lower San Joaquin Valley, California. Thermal infrared imagery appeared to be of greatest value in water table detection because of unique soil and water thermal characteristics. In particular, the flux of heat through soils underlain by shallow water tables is different from that through dry soils. Consequently, a diurnal surface anomaly occurs that appears warmer at night and cooler during the day than surrounding areas that do not have perched water tables (Estes and Simonett, 1978).

Bulk Density and Porosity

The bulk density of soil cores or clods obtained from the field may be determined in the laboratory using techniques described in a previous section. For in-situ measurements, a technique developed by soil engineers is the excavation method, also called the sand cone method, described by Blake (1965). In this method, a hole is excavated to a desired depth. The hole is filled with sand from a container using a double-cone arrangement. Ottawa sand is commonly used because the bulk density is accurately known. Thus, knowing the mass of sand to fill the cavity, the associated volume is determined by calculation. The mass of soil removed from the hole is determined by oven drying. The bulk density of the soil is then determined by dividing the dry soil mass by the calculated volume.

Alternatively, a balloon is placed in the cavity and subsequently filled with water until the cavity is completely filled. The volume of water to fill the hole is measured. The soil excavated from the hole is oven-dried and weighed. The dry mass of the soil divided by the volume is equal to the bulk density.

An indirect method for determining bulk density in-place is the gamma radiation method. Two variations of the method are used: (1) the two-hole transimission technique, and (2) the scattering technique. Sources for gamma photons include cobalt-60 and cesium-137. Sodium iodide detectors are commonly used. Details of the two-hole transmission method are described by

Blake (1965). Basically, the technique entails lowering probes containing a source of gamma radiation in one of two parallel tubes (see Figure 11). Simultaneously, a probe with a radiation detector is lowered in a second tube located a fixed distance from the first tube. In this method, the density is measured in a collimated beam a few centimeters in vertical extent.

Keys and MacCary (1971) discussed the principles of the scattering technique. In practice, a probe containing both a source and a detector is lowered into a well. The source and detector are separated by shielding. In contrast to the transmission method, the scattering method examines density in a spherical volume. According to Paetzold (1979), the scattering method requires a source of higher strength but it is not as accurate as the transmission method. Unfortunately, the double-tube transmission method is primarily suitable for determining bulk density in shallow soils rather than deeper regions because of the difficulty of installing precisely parallel tubes. Consequently, the scattering method is used to measure bulk density in the vadose and groundwater zones.

An alternative technique involves placing a source of gamma photons on the soil surface and measuring the density in an underlying hemispherical volume.

Inasmuch as bulk density measured in the field also measures the density of water present in the medium, a correction is necessary to convert the measured wet bulk density to a dry weight basis. In particular, a calibration curve is prepared by measuring count rates in a number of wet soils and subsequently determining the corresponding wet bulk density using laboratory techniques described above (e.g., the calcium carbide method).

As discussed in the next paragraph, water content values of the soil in the field are determined at the same time that a count rate is obtained. Using the calibration curve and knowing the water content, the following relationship is used to determine the dry weight bulk density (Blake, 1965):

$$D_b = D_{bw}/(1 + P_w/100) \tag{6}$$

where D_b is dry weight bulk density and D_{bw} is wet weight bulk density.

The water content of the soil may be determined by collecting field samples and oven drying or using the neutron logging method described later. (When the latter method is employed, water content is determined on a volumetric basis and the above equation must be modified.) Corey, Peterson, and Wakat (1971) described a method for simultaneously measuring soil density and water content by means of a dual source containing americium-241 and cesium-137. Commercial units with dual sources are now available.

Once the bulk density has been determined, the total porosity may be calculated from the dry weight bulk density using the relationship given in equation 2, i.e.,

$$S_t = 100\ [1 - (D_b/\rho_b)]. \tag{2}$$

Water Content by Moisture Logging

For many years, agriculturists have employed the principle of neutron moderation or thermalization to measure the volumetric water content of soils in-situ. Recently, the technique has been used by hydrogeologists to monitor water storage in the intermediate zone, particularly to delineate perching layers and mounds (Keys, 1967) and also to estimate flow rates. Figure 9 is a sequence of moisture logs from a well near the Santa Cruz River in Tucson showing the growth and dissipation of mounds during an ephemeral recharge event.

Principle of the Neutron Moderation Method--

The method of water content evaluation by neutron moderation depends on two properties relating to the interaction of neutrons with matter: scattering and capture (Gardner, 1965). High-energy neutrons emitted from a radioactive source may be slowed down or thermalized by collisions with atomic nuclei. The statistical probability for such thermalization depends on the "scattering cross section" of various nuclei. It turns out that the scattering cross section of hydrogen causes a greater thermalizing effect on fast neutrons than occurs with many elements commonly found in soils. This forms the basis for detecting the concentration of water in a soil (van Bavel, 1963). The second property of interest in the neutron moderation method is capture of slow neutrons by elements present in the soil with the release of other nuclear particles or energy. Cadmium and boron have extremely large capture cross sections compared with hydrogen. The property of energy release during capture serves as a means of detecting the concentration of slow neutrons.

In operation, when a source of fast neutrons is lowered into a soil through a suitable well or casing, a cloud of thermalized neutrons is established (Gardner, 1965). This cloud reflects the moderating effect of scattering cross sections of nuclei in the soil mass on fast neutrons. If a suitable calibration is made to isolate the moderating effects of soil nuclei other than hydrogen, changes in the volume of the thermalized cloud will reflect changes in water content. In general, the wetter the soil, the smaller the volume of thermalized neutrons. Finally, a detector which relies on capture of thermalized neutrons may be used, in conjunction with suitable electronic circuitry, to measure the water content (on a volume basis).

Instrumentation--

Instrumentation used to measure water content by neutron thermalization requires three principal components: (1) a source of fast neutrons, (2) a detector of slow neutrons, and (3) an instrument to determine the count rate from the detection equipment (Figure 10).

The first moisture loggers on the market relied on radium-beryllium as a source of fast neutrons. Because of radiation health problems with radium-beryllium, bulky shielding was necessary. A common source in units constructed today is americium-beryllium. Smaller units, such as those used by agriculturists, contain sources in the millicurie range (e.g., 50, 100, 200

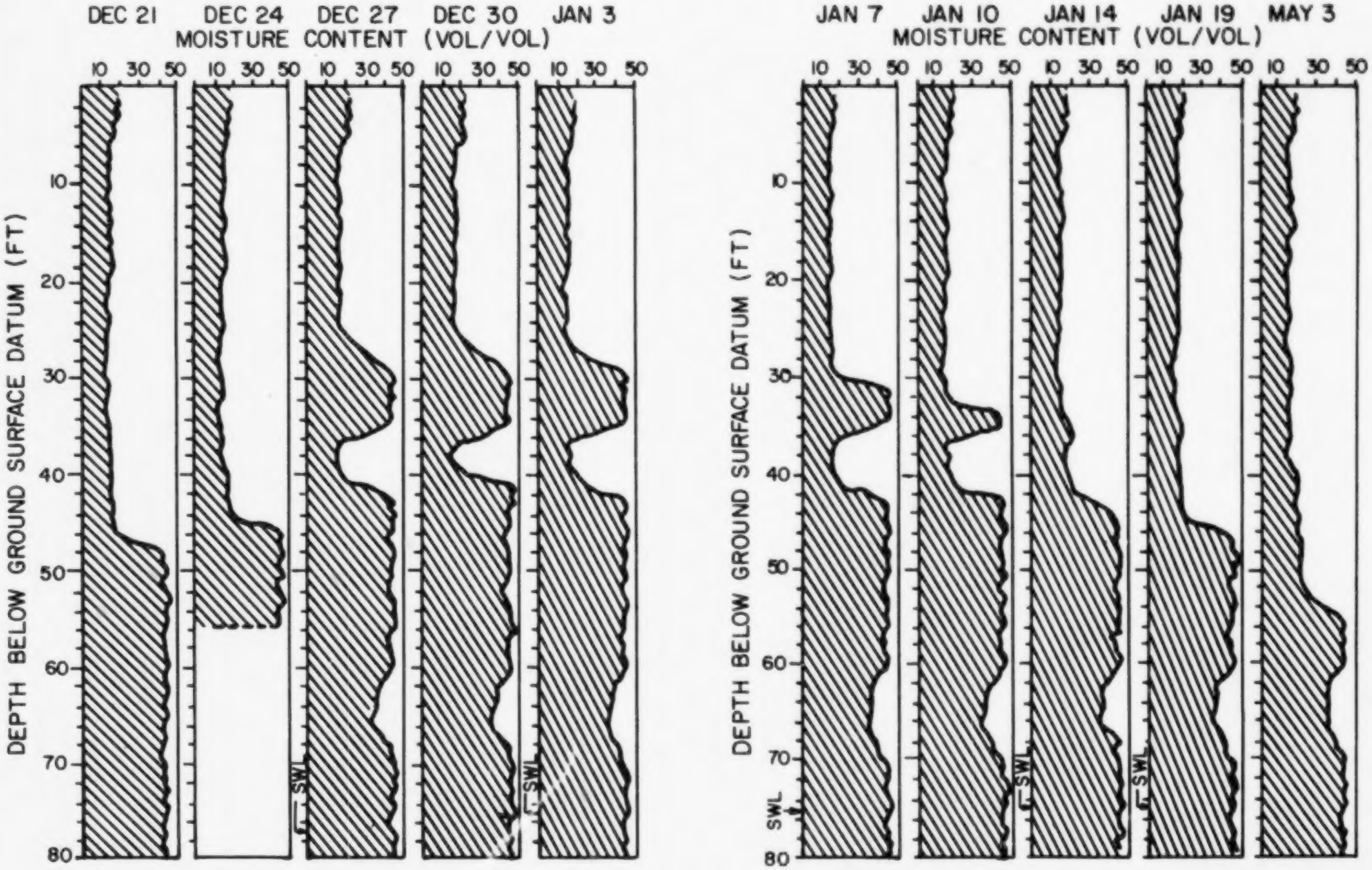


Figure 9. Sequence of water content profiles showing the growth and dissipation of perched groundwater during recharge from an ephemeral stream (after Wilson and DeCook, 1968).

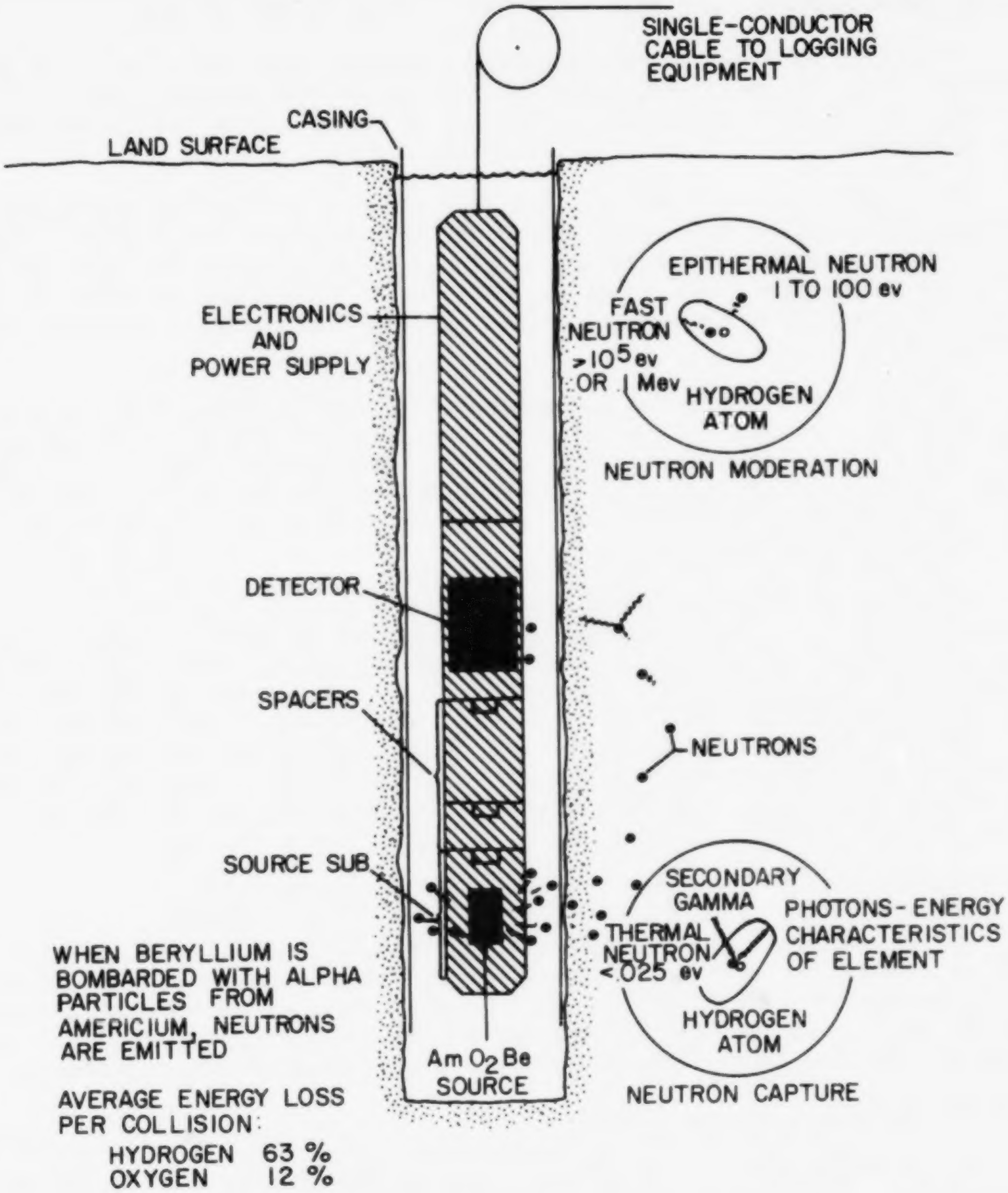

SINGLE-CONDUCTOR CABLE TO LOGGING EQUIPMENT
CASING
LAND SURFACE
ELECTRONICS AND POWER SUPPLY
EPITHERMAL NEUTRON 1 TO 100 ev
FAST NEUTRON >10^5 ev OR 1 Mev
HYDROGEN ATOM
NEUTRON MODERATION
DETECTOR
SPACERS
NEUTRONS
SOURCE SUB
SECONDARY GAMMA
PHOTONS-ENERGY CHARACTERISTICS OF ELEMENT
THERMAL NEUTRON <.025 ev
HYDROGEN ATOM
WHEN BERYLLIUM IS BOMBARDED WITH ALPHA PARTICLES FROM AMERICIUM, NEUTRONS ARE EMITTED
Am O2 Be SOURCE
NEUTRON CAPTURE
AVERAGE ENERGY LOSS PER COLLISION:
HYDROGEN 63 %
OXYGEN 12 %

Figure 10. Equipment and principles of neutron moisture logging (after Keys and MacCary, 1971).

millicuries) necessitating small-diameter casing (e.g., 2-inch). Larger units with multicurie sources, such as those used in the petroleum industry for porosity determination, are used to log wells with larger casings.

Detectors of slow neutrons comprise materials with high capture cross sections (e.g., boron). A charged particle is emitted during capture which can then be detected by a solid or gaseous counting device (van Bavel, 1963). The predominant detector is lithium-enriched BF_3.

Both the source and detector are located in a cylindrical tool which is lowered via a cable into access tubing either by hand or motor drive. The depth of measurement is determined either by graduations on the cable or by some type of counter. The relative positions of source and detector in the design of the tool are important. This question was reviewed in detail by van Bavel (1963).

The pulse emitted during the nuclear reaction resulting from capture is amplified by a preamplifier within the down-hole tool and sent through the cable to an above-ground meter for counting. Two portable meters are commonly used: rate meters or scalers. Gardner (1965) discussed the relative advantages of rate meters and scalers. Count rate is converted to water content via suitable calibration curves.

The type of meter available for soils work usually involves placing the down-hole tool at a discrete depth, taking a count or number of counts, and then moving the tool by hand to another discrete depth. This process is not troublesome for shallow soils studies. However, for deeper access wells in the intermediate zone, an inordinate amount of time would be involved. A motorized unit is available on the market which permits lowering the tool in the well at a constant rate via a motor drive. Concurrently, internal electronic components translate pulse rate into water content which is recorded on a recorder.

Calibration--

The individual neutron moisture logger should be calibrated against samples of known water content. Such calibration may be done in the field by taking a count at a given soil depth followed by a determination of the water content of soil cores from the same depth. Alternatively, calibration standards can be used, e.g., laboratory soil samples of known water content.

Field Installation of Access Wells--

Access tubes for neutron moisture logging are usually constructed of seamless steel or aluminum tubes. Plastic wells may cause difficulty in that hydrogen or chlorine atoms in the tubing may moderate the thermal neutrons, interfering with soil moisture evaluation. Aluminum wells may deteriorate in saline wells.

The inside diameter of wells should be as close as possible to the outside diameter of the probe. Work by Ralston (1967) showed that for a 100-mc Am-Be source in a 1.5-inch tool, water content could not be accurately evaluated in wells greater than 4 inches. Brakensiek, Osborn, and Rawls (1979) recommend that the radial air gap should not exceed 0.02 inch.

Wells installed for shallow water content monitoring can be easily installed by successively augering within the tube and driving the tube. Myhre, Sanford, and Jones (1969) reported on a simple power-driven auger for installing wells to about 5-foot depths.

For deeper wells, it will be necessary to use standard drilling techniques, e.g., core drilling, rotary, etc. During installation by drilling techniques, it is essential that a tight fit be established between the cavity and well to minimize the amount of vertical leakage of water. Drilling mud is not recommended for the drilling operation because it inhibits caving around the well and also interferes with water content observations. Some workers have attempted to backfill the gap between the cavity wall and the casing with fine material (Brackensiek, Osborn, and Rawls, 1979) or by grout. The results of field studies by Halpenny (personal communication, 1979) showed that bound water within grout markedly attenuates the flux of fast neutrons from a source. Consequently, the sensitivity of a logger in detecting water content changes in the surrounding formations is correspondingly reduced.

Advantages and Limitations of Neutron Moisture Logging--

The principal advantage of neutron moisture logging is that water content profiles are obtained in-situ without disturbing the soil. Thus, a history of profiles can be established at the same site during a monitoring program. Moisture logs clearly show the presence of perched water tables, together with their growth and dissipation (see Figure 9). Water content changes at a given depth in a succession of wells may provide clues on lateral flow velocities. Moisture logs may also be used to estimate unsaturated flux during drainage (see below).

Certain limitations should be noted. The presence in excessive concentrations of other fast neutron moderators (e.g., boron, chlorine) may cause erroneous water content determinations. Neutron moisture logs indicate only the water content of soils and nothing about the energetics of soil water. Water content values must be translated to equivalent soil-water suction values via a soil-water characteristic curve. Another related point is that water may move through a specific subsurface horizon without causing a change in storage. Consequently, a neutron log may not manifest water movement in this case.

Nonuniform conditions (e.g., stones or cracks) reduce the reliability of the method (Gairon and Hadas, 1973). Caution is required in handling the nuclear source and equipment is very expensive (Gairon and Hadas, 1973). Since water content is measured in a soil volume of undetermined size, it is not possible to relate water content directly to a specific sampling depth (i.e., the resolution is poor).

Water Content Via Gamma Ray Transmission

A second technique using radioisotopes to indirectly estimate water content involves gamma ray transmission, the technique discussed earlier for determining soil density. Principles of the method for water content evaluation are reviewed by Gardner (1965), Bouwer and Jackson (1974), and Reginato and van Bavel (1964). As stated by Gardner (1965): "The degree to which a beam of monoenergetic gamma rays is attenuated ... in passing through a soil column depends upon the overall density of the column. If the density of the soil less its water content is constant, then changes in attenuation represent changes in water content."

As indicated in the discussion of the use of the method for determining bulk density, the basic components for the method include a source of monoenergetic gamma rays such as cesium-137 and a detector such as NaI (TI) scintillation crystal. Accessories include a high-voltage supply, amplifier, scaler, timer, spectrum analyzer and pulse height analyzer, and a photomultiplier tube (Bouwer and Jackson, 1974).

For field usage, two parallel access wells are required, one for the source and one for the detector (see Figure 11). Both source and detector must be properly positioned with respect to each other during sampling.

Advantages and Limitations of the Gamma Transmission Method--

According to Brakensiek, Osborn, and Rawls (1979), advantages of the method include the following:

> Measurements ... by gamma-ray attenuation provide a nondestructive method of determining soil moisture and soil density. The site sampling can be repeated as often as desired. The measurements can be made vertically as close as one inch. Measurements at one inch from an interface (surface) are valid. The precision is high and accuracy of the results is excellent. Errors in measurement can usually be detected and corrected at time of measurement. No particular health hazard is involved in this type of measurement provided simple safety precautions are observed and the radioactive source is shielded when out of the access tube.

Disadvantages of the method include: changes in bulk density in shrinking and swelling materials may affect calibration, equipment is expensive, and instabilities in count rate may occur (Reginato and Jackson, 1971; Reginato, 1974). Also, the need to maintain access wells a constant distance apart is impractical for deep sampling, particularly in rocky materials.

Water Content Using Tensiometers

Tensiometers are used to measure (negative) soil-water pressures during unsaturated flow. If caution is exercised, they may be used to estimate soil-water content via suitable soil-water characteristic curves. In particular, knowing the soil-water pressure in the field via tensiometer data, a

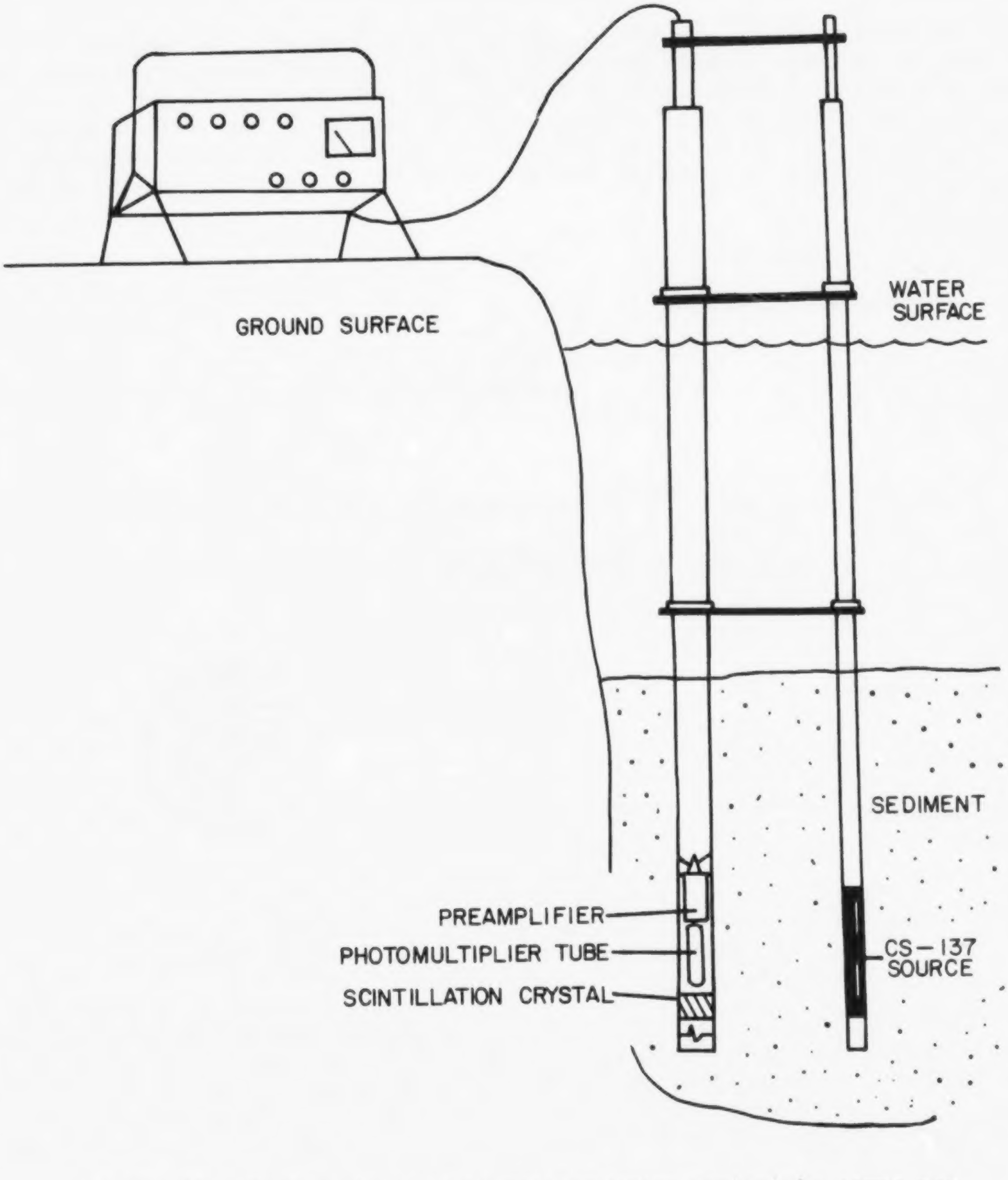

WATER SURFACE
GROUND SURFACE
SEDIMENT
PREAMPLIFIER
PHOTOMULTIPLIER TUBE
SCINTILLATION CRYSTAL
CS-137 SOURCE

Figure 11. Dual probe used to measure water content by gamma ray attenuation (after Brakensiek, Osborn, and Rawls, 1979).

soil-water characteristic curve (e.g., Figure 6) for the field soil could be consulted to determine the corresponding water content value.

Basically, a tensiometer consists of a porous ceramic cup cemented to a rigid plastic tube, a small-diameter (spaghetti) tubing leading to a manometer and terminating in a reservoir of mercury, and a filler plug in the rigid plastic tube (see Figure 12). Except for the portion of the small-diameter tubing filled with mercury, the internal volume of the system is completely filled with water. Alternative designs use bourdon type gages in lieu of mercury manometers.

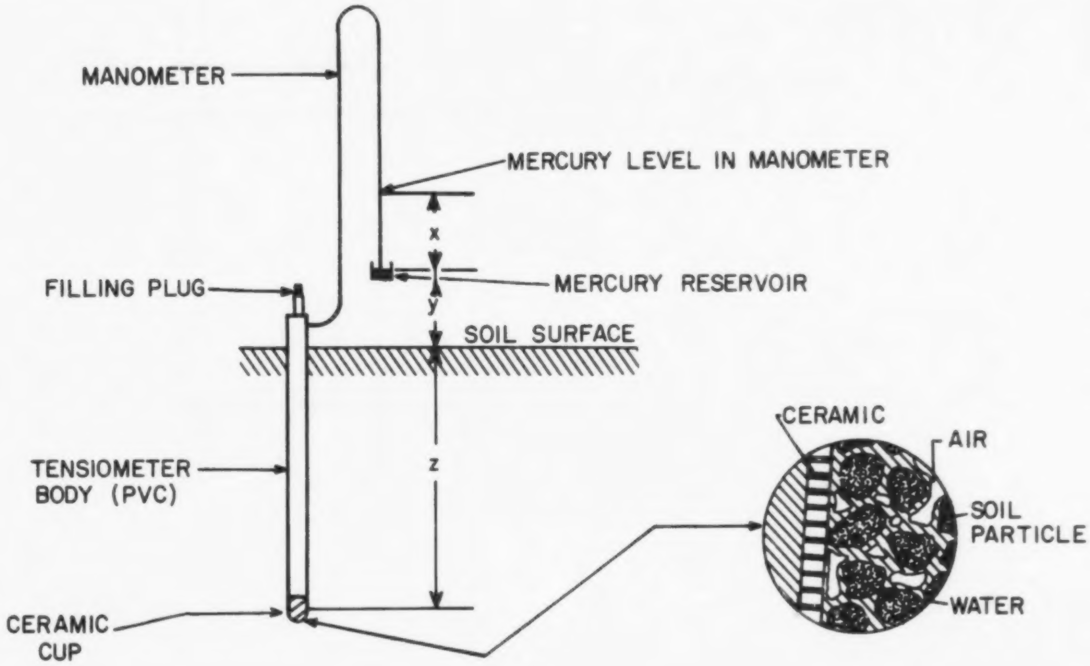

MANOMETER
MERCURY LEVEL IN MANOMETER
x
FILLING PLUG
MERCURY RESERVOIR
y
SOIL SURFACE
z
TENSIOMETER BODY (PVC)
CERAMIC CUP
CERAMIC
AIR
SOIL PARTICLE
WATER

Figure 12. Schematic representation of tensiometer and section through the ceramic cup (after Nielsen, Biggar, and Erh, 1973; and Bouwer, 1978).

When properly installed in the soil, the pores in the cup will form a continuum with the pores in the soil. Water will move either into or out of the tensiometer system until an equilibrium is attained across the ceramic cup. The mercury level in the manometer tubing will correspondingly adjust. From Figure 12, the soil-water pressure head, h, will be (Nielsen, Biggar, and Erh, 1973):

$$h = -(12.55\,x - y - z) \qquad (7)$$

where x = height of mercury column

y = distance from soil surface to mercury reference level

z = depth below land surface to ceramic cup.

Holmes, Taylor, and Richards (1967) and Brakensiek, Osborn, and Rawls (1979) noted the precautions to be taken during installation of tensiometers. The individual units should be filled with deaired water and the cups should be immersed in water during transport to the field. A hole slightly larger in diameter than the tensiometer is dug to the desired depth. A slurry is prepared from native soil and poured into the bottom of the hole. The tensiometer cup is then forced into the slurry and the tube backfilled with soil. Care must be exercised to ensure that surface water does not leak down along the tube.

Holmes, Taylor, and Richards (1967) pointed out that to ensure minimum effect of the tensiometer on the soil-water system, the sensitivity of the unit should be as large as possible. Furthermore, the response lag of the system should be kept to a minimum, particularly in rapidly changing soil-water systems.

Bianchi (1967) presented the design of a tensiometer coupled with strain gages, in lieu of the mercury system, to permit the conversion of soil-water pressure into the equivalent electrical resistance. His system would allow deep emplacement of tensiometers and the use of automatic recording equipment. Watson (1967) presented a design for a tensiometer-pressure transducer system which would also allow for the recording of soil-water pressure at depth. Figure 13 illustrates this system. He recommended such a device for recharge basin studies, and indicated that the unit can also be used for automatic recording of soil-water pressures at remote stations. In lieu of employing one transducer per unit, Brakensiek, Osborn, and Rawls (1979) described a strain gage/recorder combination using one transducer to service several tensiometers in turn.

Advantages and Limitations of Tensiometers for Measuring Water Content--

The principal advantage of tensiometer units is that they provide in-place continuous readings of soil-water pressure, which may be translated into water content values (Gairon and Hadas, 1973). Two limitations of tensiometers are (1) they are useful only in measuring soil-water pressures in excess of about -0.8 bar, and (2) they are sensitive to temperature. The most serious problem with the method is that, because of hysteresis, it is necessary to know whether the soil is wetting or drying when the tensiometer is read. In particular, during the redistribution of soil water following infiltration, portions of the profile may be wetting while others are drying. Thus, a tensiometer reading could represent the value of a wetting curve, a drying curve, or one of an infinite number of scanning curves.

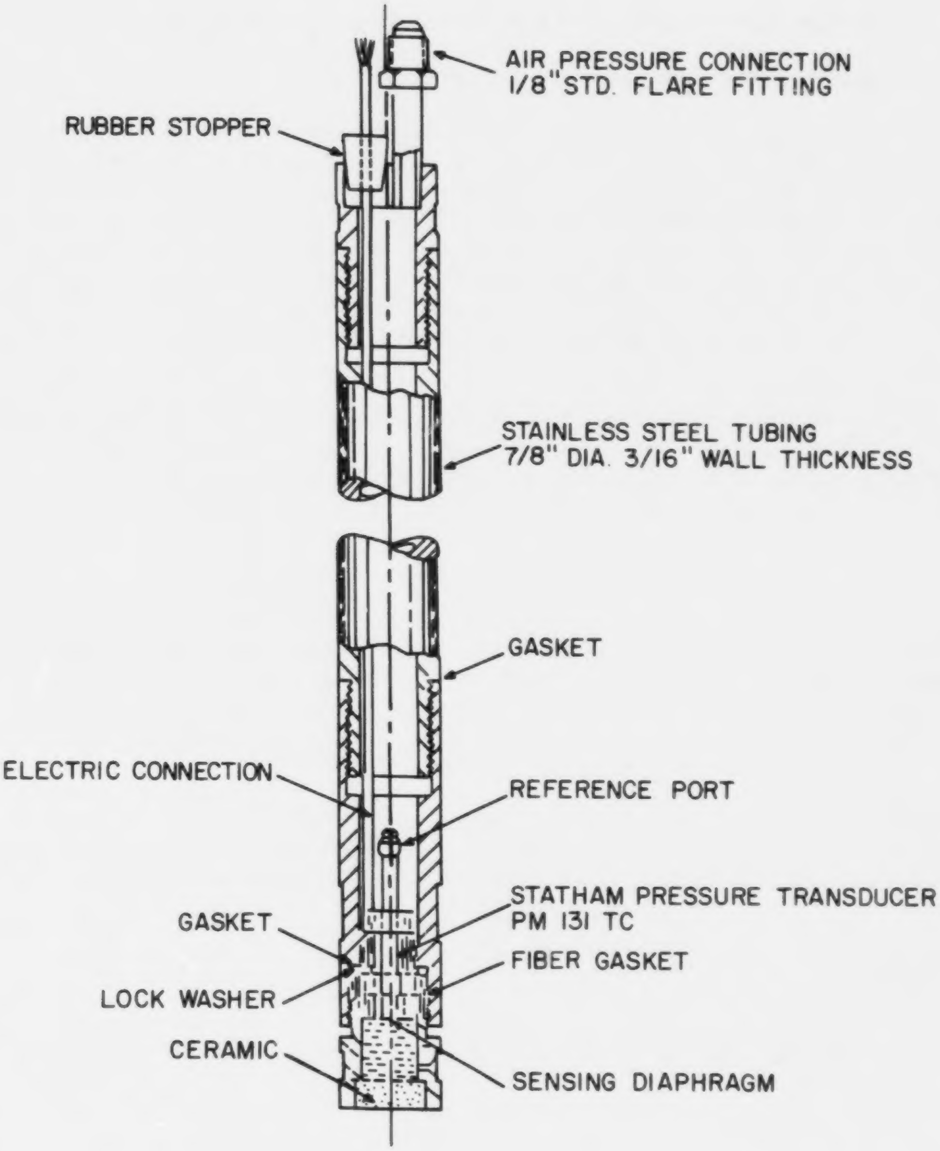

AIR PRESSURE CONNECTION
1/8" STD. FLARE FITTING
RUBBER STOPPER
STAINLESS STEEL TUBING
7/8" DIA. 3/16" WALL THICKNESS
GASKET
ELECTRIC CONNECTION
REFERENCE PORT
STATHAM PRESSURE TRANSDUCER
PM 131 TC
GASKET
FIBER GASKET
LOCK WASHER
CERAMIC
SENSING DIAPHRAGM

Figure 13. Cross section of tensiometer-pressure transducer assembly (after Watson, 1967).

Water Content Using Electric Resistance Blocks

Electrical resistance blocks, used to measure either soil-water content or soil-water pressure, consist of electrodes embedded in suitable porous material. Plaster of Paris, fiber glass, and nylon cloth have been used. The principle of operation of these blocks is that water content (or negative pressure) within the blocks responds to the water content (or suction) of the soil with which the blocks are in intimate contact (Holmes, Taylor, and Richards, 1967). Correspondingly, the electrical properties of the block also change as reflected by measurement of resistance.

Moisture blocks are calibrated in soil from the site at which they are to be installed. Such calibration involves evaluating resistance readings against a range of soil-water contents or negative pressures.

Phene, Hoffman, and Rawlins (1971) presented an alternative to the standard gypsum block comprising a heat source and sensor embedded in a ceramic body (see Figure 14). This device measures both soil-water content and matric potential. The basis of the units is that air is a good thermal insulator with respect to water. With drying and entry of additional air into the media, the remaining water films become thinner. Consequently, the flow path for heat conduction increases and a larger temperature gradient is needed to dissipate heat. Phene, Rawlins, and Hoffman (1971) also discussed laboratory and field tests with their sensor. Apparently, the accuracy of the sensor was as good as or better than other techniques used to measure matric potential.

Advantages and Limitations of Gypsum Blocks--

Holmes, Taylor, and Richards (1967) discussed the advantages of resistance blocks indicating that (1) they appear to be best suited for general use in study of soil-water relations, (2) they are inexpensive, and (3) they can be calibrated for either suction or water content. Generally, blocks are used for soil-water pressures less than -0.8 atmosphere.

Phene, Hoffman, and Rawlins (1971) reviewed the problems with gypsum blocks together with alternatives to circumvent these problems:

> In a gypsum block the water content is measured by the electrical resistance between two imbedded electrodes. Several problems are usually encountered when using gypsum blocks; first, salinity affects electrical conductivity independently of water content; second, the gypsum used in attempting to mask variations in soil salinity eventually dissolves resulting in an unstable matrix for the sensor; third, contact resistance between the porous body and the soil can restrict the exchange of water between the block and the soil; and fourth, the hysteresis in the water content-matric potential relation of the porous body can cause errors in the interpretation of the data depending on whether the measurements are taken during a drying or wetting cycle. Elimination of the first problem requires a measurement of the water content of the porous body that is independent of salinity. This, in turn, would eliminate the need

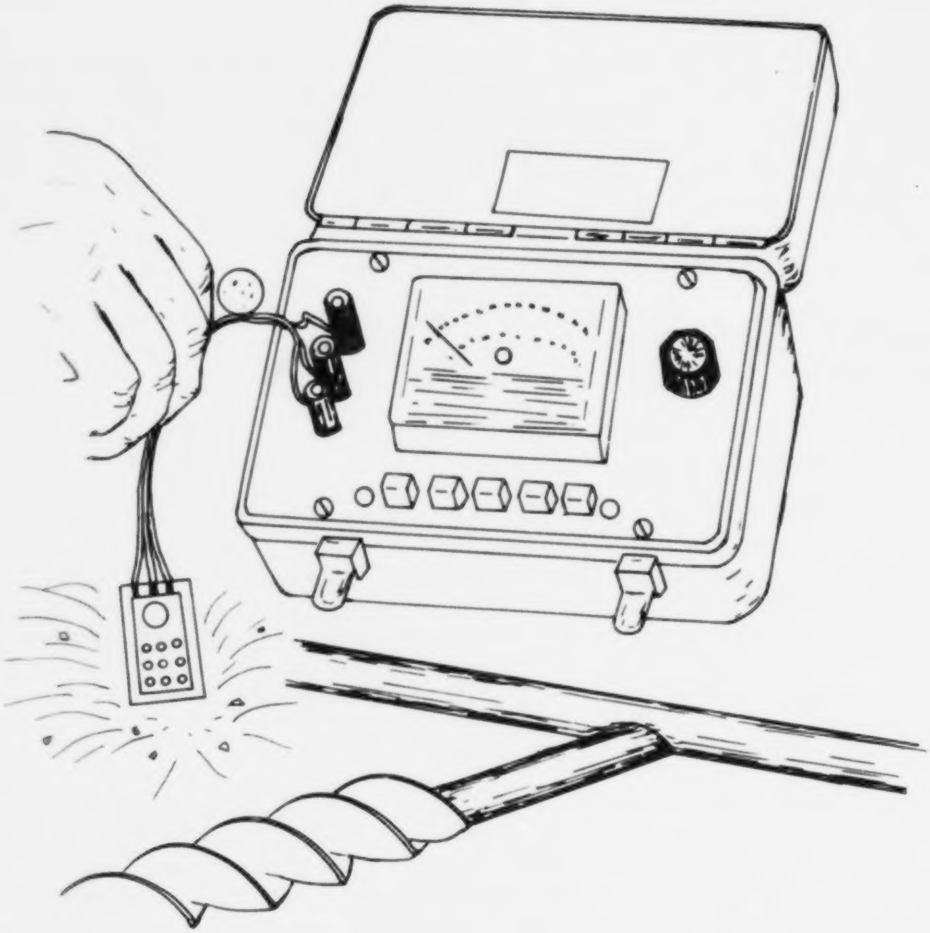


Figure 14. Thermistor soil cell, meter, and hand auger. Cell is being placed in a soil cavity (after Soiltest Inc., 1976).

for the buffer material so that a stable matrix could be used. Contact resistance between the block and the soil can be minimized by packing the dry soil carefully around the sensor and then irrigating. Generally, the status of soil water is only of interest during drying conditions, since after a rain or an irrigation, the soil water content changes rapidly as the wetting front passes. Thus, hysteresis is of little concern and only a desorption calibration curve is needed.

Blocks are rather insensitive to moisture changes in the wet range (Gairon and Hadas, 1973). Hence, blocks are commonly used as an adjunct to tensiometers to monitor moisture content values at pressures less than -0.8 atmosphere.

Water Characteristic

As discussed by Bouwer (1978), the negative pressure head in the vadose zone equals the vertical distance above a water table provided vertical flow is not occurring. Consequently, for uniform conditions, a plot of the change in volumetric water content with distance above a water table represents the water characteristic curve of the vadose zone material, again assuming that there is no vertical flow. In practice, the volumetric water content could be determined using the neutron moisture logger described previously.

Bouwer (1978) described the nature of characteristic curves for layered conditions. The negative pressure head still equals the vertical distance above the water table. However, the water content-head relationship depends on the soil at the measurement point. In other words, an irregular water content distribution occurs above the water table, indicating that although head changes occur continuously with vertical distance, discontinuities may occur in water content distribution. Bouwer (1978) indicated that certain fine-textured soils may actually be saturated (i.e., contain perched groundwater lenses), whereas coarser-textured material above and below may be unsaturated.

Specific Yield, Fillable Porosity, and Specific Retention

A number of techniques are available for estimating the specific yield of the groundwater zone. If the storage properties of material above and below the water table are similar, the specific yield value determined for a groundwater zone could be assumed to approximate the specific yield in the vadose zone. In actuality, this assumption may be very tenuous because of the marked variations in lithology noted in the field.

The most common technique for estimating specific yield is to conduct pumping tests on wells. (Such tests also provide data on the transmissivity, T, of the aquifer.) Common testing techniques are described by Lohman (1972). Alternatively, if long-term groundwater withdrawals are known together with the corresponding head changes, specific yield values of the regional groundwater system could be estimated (e.g., Matlock and Davis, 1972).

As indicated by Stallman (1967) and Bouwer (1978), specific yield values determined from short-term pumping tests may underestimate the true value because of delayed drainage. Instead of relying on results from short-duration tests, Stallman (1967) suggested that a more realistic estimate of specific yield could be obtained by observing variations in water content profiles during the decline of a water table. In particular, the specific yield could be calculated by using a sequence of water content profiles to determine the volume of water drained near the water table during a decline in the water table (Bouwer, 1978). Meyer (1963) used this technique to estimate temporal variations of the apparent specific yield near pumping wells.

The use of neutron moisture logs appears to be the most suitable approach for estimating the specific yield of vadose zone sediments near the water table. Such logs could also be used to estimate the specific yield of deposits in other regions of the vadose zone where perched groundwater bodies are generated during cyclic recharge events.

Neutron moisture logs also afford the means of estimating the fillable porosity of sediments near the water table, i.e., the volume of water placed into storage per unit rise in water level. As indicated in another section, because of entrapment of air during the rise of a water table, the fillable porosity will initially be less than the specific yield.

The specific retention may also be determined from neutron moisture logs obtained during the recession of a water table. It will be recalled that the specific retention equals the volume of water retained against the pull of gravity during drainage.

UTILIZATION OF MONITORING METHODS FOR OBSERVING STORAGE CAPACITY BENEATH A POLLUTION SOURCE

The methods described in this section are examples of the available technology for estimating the capacity of a given volume of vadose zone underlying a pollution source to store water and water-borne pollutants. The example given in the section entitled "Thickness" showed the maximum (theoretical) capacity. In light of the discussion on field capacity, however, it is apparent that the actual storage capacity would be less than the maximum because of drainage. Another example using equation 1 will illustrate this point.

Repeating equation 1:

$$d_w = \frac{\Delta P_w}{100} \frac{D_b}{D_w} d_{vz}.$$

In practice, bulk density values (D_b) required to solve equation 1 could be evaluated using the gamma ray transmission method for shallow depths and the scattering method for deeper regions. Water content values under background conditions and at field capacity could be estimated from such alternative methods as neutron moisture logging, gamma ray transmission or scattering, tensiometers, and moisture blocks.

Assume again the following conditions: (1) 100-foot thick vadose zone, (2) bulk density of 8.1 lb/ft^3 ($1.3\ gm/cm^3$), and (3) water content of dry vadose zone material = 10 percent of weight. Assume also that the water content of the material comprising the vadose zone at field capacity is 20 percent by weight.

From equation 1:

$$d_w = \frac{20 - 10}{100} \times 1.3 \times 100$$
$$= 13 \text{ feet.}$$

For this hypothetical case, 13 acre-feet of water could be theoretically stored in the vadose zone per acre of surface area when the media is at field capacity.

SECTION 5

MONITORING WATER MOVEMENT IN THE VADOSE ZONE

In Section 4, methods for estimating the capacity of the vadose zone to store water and water-borne pollutants were reviewed. Although a knowledge of storage capacity is important, individuals charged with monitoring water-borne pollutants are more likely to be concerned with the rate of movement through the vadose zone. In other words, how fast are water and entrained constituents moving from a surface source to the water table? In this section, the movement of water alone is examined. Relationships affecting the mobility of pollutants are reviewed in Section 6.

In practice, it is a fairly simple task to measure the rate of water movement across a soil-water interface at the land surface. For example, seepage in a pond can be estimated by determining water level changes and using appropriate depth-volume-surface area relationships. In contrast, the process of water movement through the soil and lower vadose zone underlying the source is of staggering complexity. Elements contributing to the complexity of flow include variations in the state of water saturation and spatial variations in the physical/hydraulic properties of the vadose zone.

In light of the difficulties in attempting to describe water movement in the vadose zone, it is not possible to present exact techniques for estimating transit time of water and water-borne pollutants through this region. However, a number of indirect methods are available which will be reviewed in this section. As in the previous section, a review of technical elements is presented first, together with laboratory techniques. Field methods are discussed in a later subsection. Finally, a possible arrangement of facilities for estimating movement in both saturated and unsaturated zones is presented.

TECHNICAL REVIEW

For ease of discussion, the following categorization will be employed: (1) infiltration at the land surface, (2) unsaturated flow in the vadose zone, and (3) flow in saturated regions of the vadose zone. This subdivision is somewhat arbitrary inasmuch as infiltration is generally regarded as an unsaturated flow process.

Infiltration

Infiltration is the process by which water enters a soil. The maximum rate at which water enters a soil is the infiltration capacity. The Soil

Conservation Service has classified soils on the basis of infiltration rates and transmission rates as follows (U.S. EPA et al., 1977):

Class	Transmission rate (in/hr)	Transmission rate (cm/hr)
Very slow	<0.06	<0.15
Slow	0.06 to 0.2	0.15 to 0.51
Moderately slow	0.2 to 0.6	0.51 to 1.5
Moderate	0.6 to 2.0	1.5 to 5.1
Moderately rapid	2.0 to 6.0	5.1 to 15.2
Rapid	6.0 to 20.0	15.2 to 50.8
Very rapid	>20.0	>50.8

The infiltration rate of soil wetted continuously is initially high but decreases steadily with time, approaching an asymptotic value. The asymptotic value is approximately that of the saturated hydraulic conductivity of the soil. The physical reason for the typical decrease in infiltration with time is described by Hillel (1971):

> When infiltration takes place into an initially dry soil, the suction gradients are at first much greater than the gravitational gradient .... As the water penetrates deeper and the wetted part of the profile lengthens, the average suction gradient decreases, since the overall difference in pressure head (between the saturated soil surface and the unwetted soil inside the profile) divides itself along an ever-increasing distance. This trend continues until eventually the suction gradient in the upper part of the profile becomes negligible, leaving the constant gravitational gradient as the only remaining force moving water downward in this upper or transmission zone. Since the gravitational head gradient has the value of unity ... it follows that the flux tends to approach the hydraulic conductivity as a limiting value.

The distribution of water as a function of depth within a soil during infiltration is generally called a water content profile. As first described by Bodman and Coleman (1944), water content profiles may be divided into three distinct zones: (1) a transmission zone in which the upper few centimeters of soil may be saturated and below which the soil is very nearly saturated--this zone continually lengthens during infiltration; (2) a wetting zone exhibiting rapid changes in water content with depth and time; and (3) the wetting front, the visible limit of water penetration. Figure 15 shows water content profiles during infiltration into a soil which is continuously flooded and into the same soil when water is applied slowly by sprinkler. Because of this slow application rate, the water content values in the profile during sprinkling at all times remain below the values for the ponded case.

Following the cessation of surface flooding during a water-spreading operation, water becomes redistributed in the soil profile. The upper portion of the profile drains, while the lower portion continues to wet. The physics of redistribution is complex, particularly because of hysteresis.

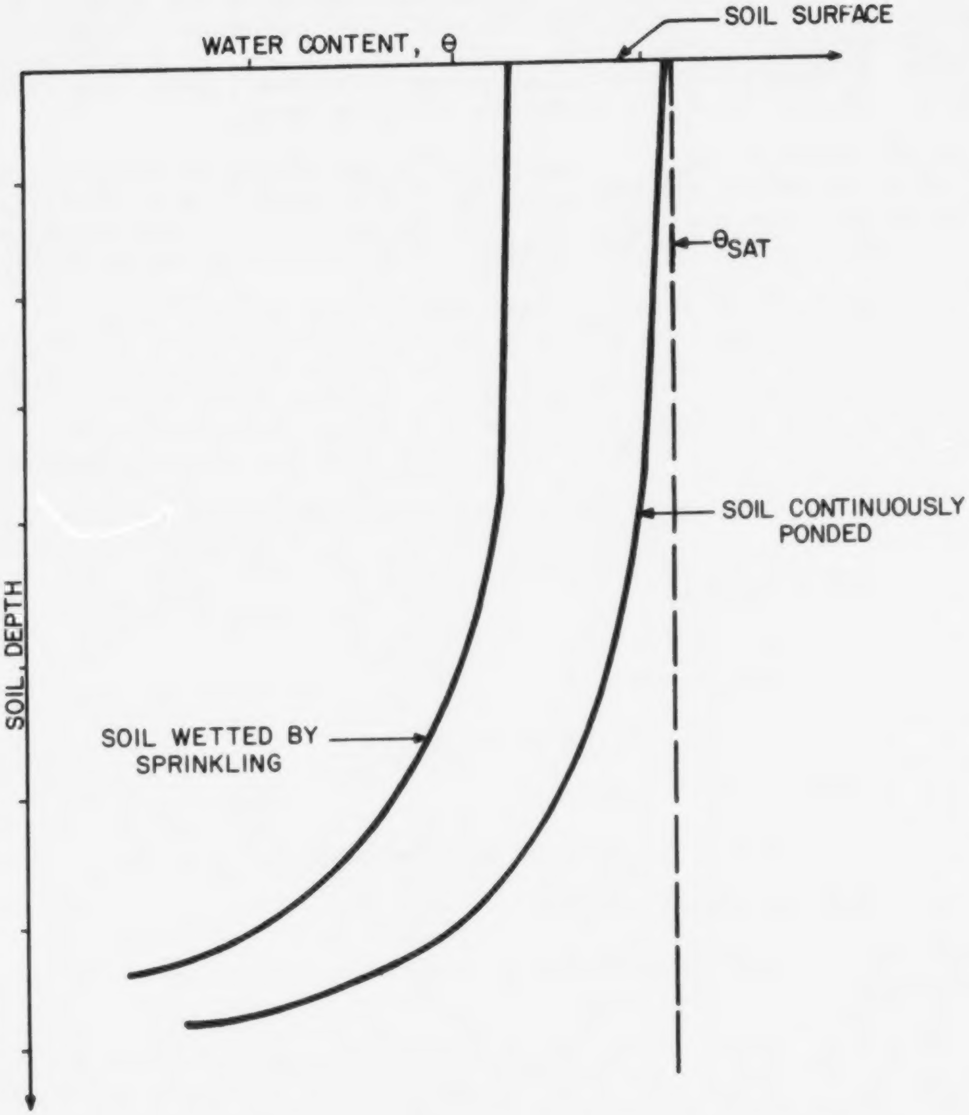


Figure 15. Water content profiles of a soil wetted slowly by sprinkling and a soil continuously ponded (after Hillel, 1971).

For years hydrologists have attempted to model infiltration using (1) algebraic and empirical equations, (2) approximations of rigorous flow equations, and (3) equations derived from simplifications of the physical system (Brakensiek, 1979). The fact that the work still continues is apparent by examining the papers presented at a recent workshop on infiltration research, published by the Sciences and Education Administration (1979). The interested researcher should consult this publication for details.

Of the algebraic/empirical equations for describing infiltration, Brakensiek (1979) recommends the Green and Ampt (1911) equation and the Philip (1969) two-parameter equation. The Green and Ampt equation was discussed in detail by Childs (1969), Philip (1969), Hillel (1971), and Bouwer (1978).

Assumptions of the Green and Ampt approach are (Hillel, 1971): "... There exists a distinct and precisely definable wetting front, and that the matrix suction at this wetting front remains effectively constant, regardless of time and position." (Matrix suction is a measure of the sorptive capacity of a soil for water.) The Green and Ampt equation has been found to be satisfactory for describing infiltration into initially dry coarse-textured soils.

As presented by Bouwer (1978), the Green and Ampt equation is as follows:

$$V_i = K \frac{H_w + L_f - h_{cr}}{L_f} \tag{8}$$

where V_i = infiltration rate

K = hydraulic conductivity of wetted zone

H_w = depth of water above soil

h_{cr} = critical pressure head of soil for wetting

L_f = depth of wetting front.

Brakensiek (1979) itemized parameter estimation procedures for the Green and Ampt equation.

Philip (1969) derived a two-parameter algebraic equation from physical principles to describe vertical one-dimensional infiltration by solving the basic partial differential equation for unsaturated flow. Philip's solution described the time dependence of infiltration in terms of a power series which, when t is not too large, reduces to:

$$I_t = St^{1/2} + At; \; v_i = 1/2\, St^{-1/2} + A \tag{9}$$

where I_t = cumulative infiltration

v_i = infiltration rate at time t

S = sorptivity

A = a constant.

According to Bouwer (1978), the sorptivity depends on the pore configuration of the soil, the initial water content, and the depth of applied water. Approaches for estimating the parameters in the Philip two-parameter equations are summarized by Brakensiek (1979). For short infiltration events, A is taken to equal K/2, where K is the saturated hydraulic conductivity. For long infiltration events, A is equal to K (Bouwer, 1978).

The initial infiltration rate of a soil is theoretically infinite (Bouwer, 1978). Note that both equations 8 and 9 comply with this requirement. In the case of equation 8, the value of the initial rate approaches infinity because the depth to the wetting front is zero ($L_f = 0$). For equation 9, the infiltration rate, V_i, at time $t = 0$ is also infinite because the first term on the right-hand side overwhelms the second (A) term.

An empirical infiltration curve commonly used by hydrologists is that of Horton. This equation may be written in the following form:

$$V_i = V_\infty + (V_o - V_\infty)e^{-Kt} \tag{10}$$

where V_i = infiltration rate at time t

V_o = initial infiltration rate

V_∞ = final infiltration rate

e = base of natural logarithms

K = a constant depending on soils and vegetation.

Values for K are found from field infiltration data by expressing equation 10 in logarithmic form. A basic problem with Horton's equation is that it does not satisfy the theoretical requirements that the initial infiltration must be of infinite value (Bouwer, 1978, p 256). According to Bouwer, the equation is best suited to describing infiltration when water is applied by rain or sprinkling for short periods.

Factors affecting the infiltration capacity (and thus the potential amount of pollutants entering the lower vadose zone) of a soil include the following: soil texture, soil structure, initial water content, presence of shallow impeding layers, entrapped and confined air, biological activity, entrained sediment, and salinity of applied water. The effect of soil texture on infiltration is complex. For example, a clay soil may posses a rather high sorptivity value but water movement is retarded by the energy losses in the fine pores. Inasmuch as sorptivity decreases with time, the long-term infiltration rate depends on the hydraulic conductivity of a soil. Thus, a sandy soil should have a greater long-term infiltration rate than a clay soil. Soil structure has a profound effect on infiltration rate, particularly at the beginning of infiltration. This is, water moves preferentially very rapidly through the larger interpedal cracks at the beginning of infiltration. Later, as the soil swells closing the cracks, water movement occurs through the soil blocks. Hillel (1971) pointed out that the long-term infiltration rate of a highly structured soil approaches the rate of a uniform soil because of the control imposed by the hydraulic conductivity of the lower soil zone.

The initial water content of a soil affects the infiltration rate because of the concurrent effect on sorptivity and available porosity. Thus, the dryer the soil, the greater will be the initial infiltration rate compared to a wetter soil (see Figure 16). The final or long-term infiltration rate of a soil, however, is independent of the initial water content. Shallow impeding layers within the soil zone retard the infiltration rate compared to a non-stratified soil. As pointed out by Hillel (1971), the effect is the same whether a fine soil is underlain by a lens of coarser material, or vice-versa. Air entrapped in the soil pores during infiltration effectively behaves as solid grains in restricting water movement. Eventually, however, the air may dissolve in the water, causing an increase in intake rates. Confined air ahead of the wetting front may also restrict infiltration because of the buildup in air pressure due to a restricting lens or shallow water table (Wilson and Luthin, 1963). In fact, the air pressure may increase to the point that the air-entry value of the soil is exceeded and air escapes at the surface.

Fine sediment entrained in the water may clog the surface pores and effectively create a barrier to the infiltration process. Water movement in the underlying pores becomes unsaturated and flow occurs primarily in the finer pore sequences. Microbial activity may also result in a clogging of soil pores with the byproducts of their metabolic processes, e.g., slimes or gasses. Microorganisms may produce a secondary effect in that the pH of the overlying water may be increased due to microbial respiration, resulting in a precipitation of carbonates within the surface pores (Bouwer, personal communication, 1978).

The salinity of the applied water may be a dominant factor in affecting infiltration for soils containing substantial clay fractions (i.e., soils with a tendency to shrink and swell). Thus, it is well known among agriculturists that a high sodium concentration in the applied water relative to calcium and magnesium may disperse the clays, slowing down intake rates. This effect is discussed in detail in the section "Saturated Flow in the Vadose Zone."

Unsaturated Flow in the Vadose Zone

The two fundamental mathematical expressions describing the flow of water in porous media are Darcy's equation and the equation of continuity. For unsaturated flow, Darcy's equation is written (after Nielsen, Biggar, and Erh, 1973) as:

$$J = -K(\theta)\nabla H \tag{11}$$

where J = specific discharge or flux (volume per unit area per unit time); also called Darcian flux

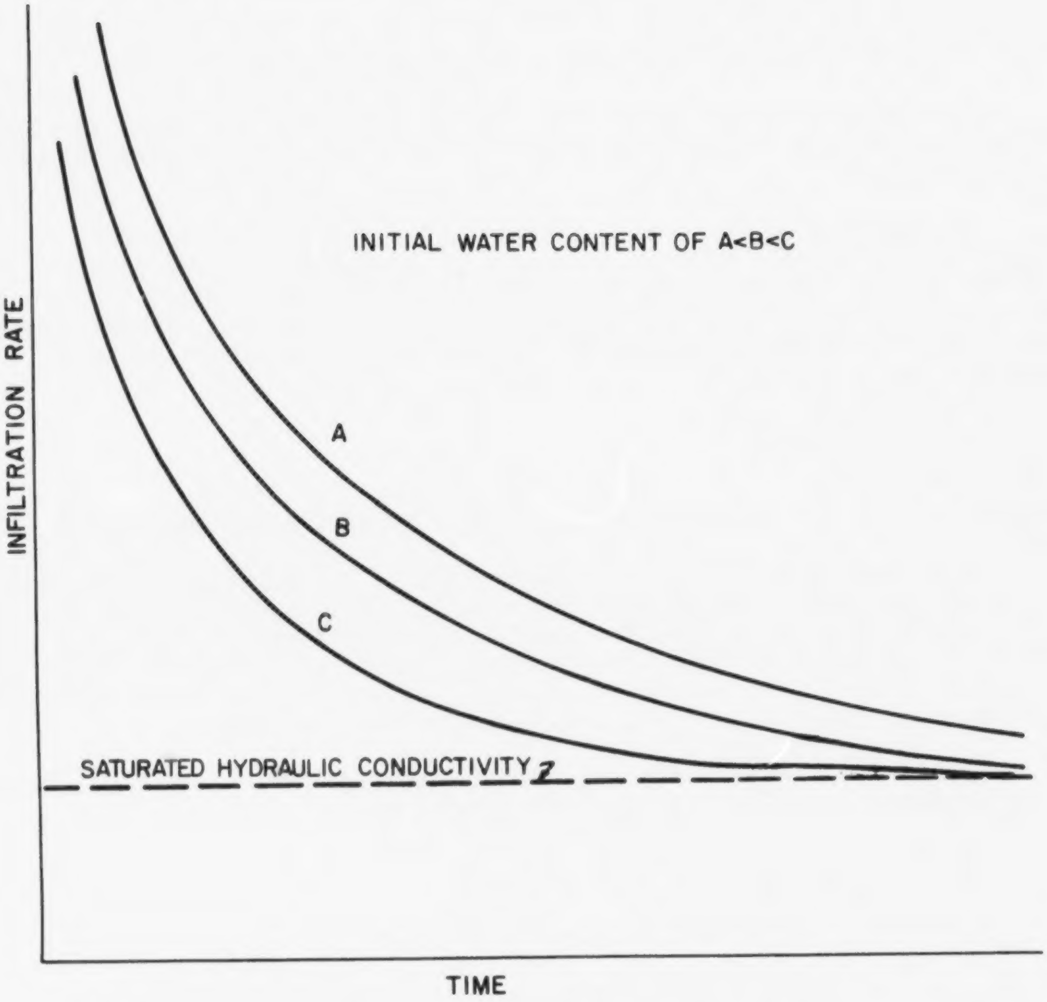


Figure 16. The effect of initial water content of a soil on infiltration rates.

$K(\theta)$ = hydraulic conductivity, expressed as a function of water content, θ

H = total hydraulic head, the sum of soil-water pressure head, h, and potential head, z

∇H = the hydraulic gradient, expressed in vector form.

The specific discharge has the units of velocity. The negative sign is used by convention because flow occurs in the direction of decreasing hydraulic head.

As expressed by equation 11, the specific discharge is a macroscopic quantity inasmuch as the cross-sectional area of flow includes both solids and voids. A more representative depiction of velocity would exclude solids from the flow path and account for the water content of the void spaces. Consequently, an alternative expression for flow in unsaturated media is as follows:

$$\bar{v} = \frac{J}{\theta} = \frac{-K(\theta)}{\theta} \nabla H \qquad (12)$$

where $\bar{v}$ is the average linear velocity (Freeze and Cherry, 1979).

As pointed out by Freeze and Cherry (1979), the average linear velocity is also a macroscopic quantity. The actual or microscopic velocity within pore sequences would be greater than $\bar{v}$ because the tortuous flow paths are greater than the linearized path assumed in defining $\bar{v}$. However, note that $\bar{v}$ will always be greater than J for a given soil, permitting a more realistic estimate of travel time of nonreactive water-borne pollutants.

The second important flow equation is the equation of continuity which in one dimension (the z direction) is written as:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left(K(\theta) \frac{\partial H}{\partial z}\right) \qquad (13)$$

where t is time.

Basically, the hydraulic conductivity is a factor representing the ease with which water moves through a soil or aquifer. In equation 12, the unsaturated hydraulic conductivity is shown to be a function of water content, θ. However, as discussed in a previous section, θ depends on the negative pressure head, h. As a consequence of the h-θ relationship, the unsaturated hydraulic conductivity is also a function of the negative pressure head. The relationship between the unsaturated hydraulic conductivity and negative pressure head is shown on Figure 17. The effect of hysteresis is also shown on the figure.

The important feature of Darcy's equation relative to the movement of water into the vadose zone beneath a disposal area is that two factors must be defined in order to characterize the flow rate. These factors are the

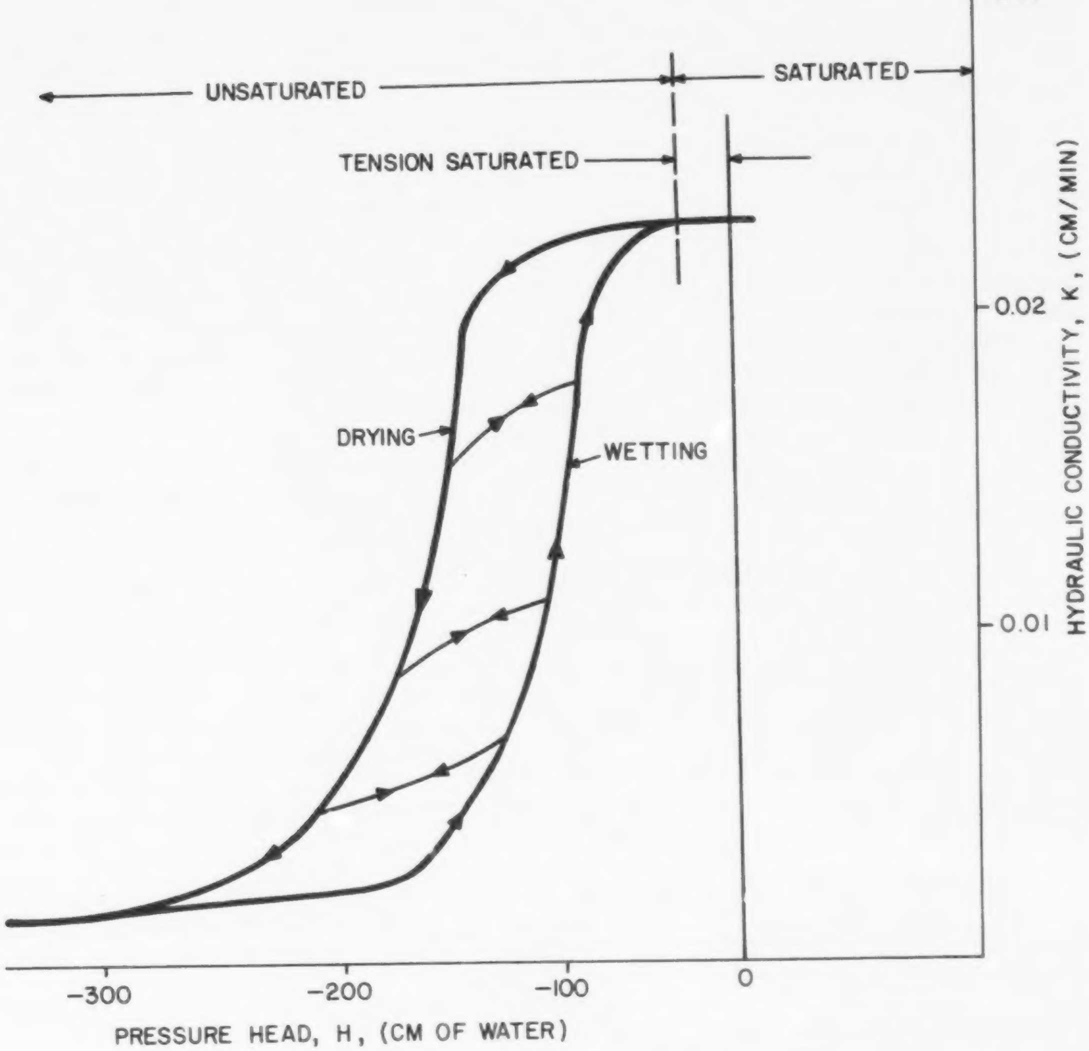


Figure 17. Effect of the negative pressure of soil water on the unsaturated hydraulic conductivity. The figure also shows the effect of hysteresis on K (after Freeze and Cherry, 1979).

hydraulic conductivity and the hydraulic gradient. Thus, the purpose of laboratory and field techniques (discussed below) is to determine K(θ), or K(h), and the hydraulic gradient.

A number of laboratory techniques are available for estimating the relationship between the unsaturated hydraulic conductivity and pressure head or water content. At the outset, it should be emphasized that because the hydraulic conductivity is dependent on water content (and thus head) in unsaturated soils, it is extremely difficult to measure K(θ) and K(h) in the laboratory or field. Thus, for unsaturated soils, the water content, θ, changes continuously along the flow path and $K(\theta)$ values also vary continuously. That is, a unique value for the hydraulic conductivity cannot be determined.

One laboratory technique for determining $K(h)$ as a function of h is the so-called long-soil column method. Briefly, the method entails introducing water at the soil surface of a long-soil column at a constant rate such that unsaturated flow is produced in the soil. The soil at the base of a column may eventually become saturated but the remainder of the soil will be unsaturated. The test is continued until the hydraulic gradient becomes equal to one or, in other words, θ is nearly constant throughout the upper region of the column. Hydraulic heads are measured via tensiometers placed in the column. When the hydraulic gradient equals one, the unsaturated hydraulic conductivity equals the flow rate divided by the cross-sectioned area from Darcy's equation. Note that the resultant hydraulic conductivity is related to the water content in the column at the time of the experiment. The experiment may be repeated at different flow rates and different water content values to provide a range of $K(\theta)$ - θ relationships.

Alternative laboratory methods for determining the unsaturated hydraulic conductivity at successive steady-state flow rates are presented by Klute (1965). These methods are suitable for soil cores. One of the most common laboratory methods, the pressure outflow technique, consists of applying successive air pressures to a moist soil section in an appropriate chamber. Using the air pressures and related water content values, an equation (Kirkham and Powers, 1972, p 274) is solved for corresponding $K(\theta)$ values. In contrast to such steady-state methods, Alemi, Nielsen, and Biggar (1976) presented centrifugal techniques for determining the unsaturated hydraulic conductivity during transient flow conditions.

Indirect methods for estimating $K(\theta)$ have relied on soil-water characteristics (for example, Millington and Quirk, 1959; and Jackson, Reginato, and van Bavel, 1965). Paetzold (1979) indicated that a computer program is available to calculate $K(\theta)$ from H versus θ curves.

Saturated Flow in the Vadose Zone

For saturated flow, Darcy's equation is written in the form:

$$v = -K\nabla H \tag{14}$$

where v = specific discharge

K = saturated hydraulic conductivity

∇H = hydraulic gradient.

To account for flow only within the pore space, Darcy's equation is also written as:

$$\bar{v} = -\frac{K}{S_t} \nabla H \qquad (15)$$

where $\bar{v}$ = average linear velocity

S_t = porosity.

As with unsaturated flow, the average linear velocity is a macroscopic quantity of lesser magnitude than the actual microscopic velocity in the pore sequences.

In the context of this report, saturated flow will be considered only in perched groundwater regions of the vadose zone. The complex flow patterns within groundwater mounds immediately above a water table will not be considered.

It is commonly assumed that perched groundwater is always created by a relatively impermeable layer which impedes vertical water movement. Although this concept is true, perched groundwater may also develop at the interface between two layered regions which individually may be quite permeable.

Figure 18, reproduced from Bear, Zaslavsky, and Irmay (1968), illustrates the requisite condition for perched groundwater formation when a region of higher permeability overlies a region of lesser permeability in the vadose zone. For the first case, the flow rate, J, is less than K_2, the hydraulic conductivity in the lower strata. Flow is unsaturated and water could not be extracted from the system except by means of a suction cup. In the second case, J equals K_2 and the soil water pressure is atmospheric. Again, it may not be possible to obtain a water sample except by means of a suction cup. In the third case, J is greater than K_2 and a region of positive water pressure is present at the interface. Water samples could be obtained from a well or open cavity.

For saturated media, the hydraulic conductivity in Darcy's equation is not generally expressed as a function of the water content. However, it is necessary to account for possible variations in the hydraulic conductivity depending on the direction of flow. For example, in layered alluvial deposits, the hydraulic conductivity in the horizontal direction is generally greater than in the vertical direction. The condition of having variable K values depending on direction is called anisotropy. In anisotropic media, the hydraulic conductivity is a second rank tensor (Bear, 1972).

Inasmuch as the hydraulic conductivity, K, of a soil is a measure of the ease of water movement, it follows that K depends greatly on soil texture and structure. Thus, a sandy soil will have a greater hydraulic conductivity than a clay soil, even though the latter has a greater porosity. Similarly, a well-structured soil will permit rapid flow through the cracks. Such a soil will have a greater hydraulic conductivity than will a poorly-structured soil, even though the soil type may be the same. Ranges of hydraulic conductivity values for representative water-bearing materials in California are shown in Table 3.

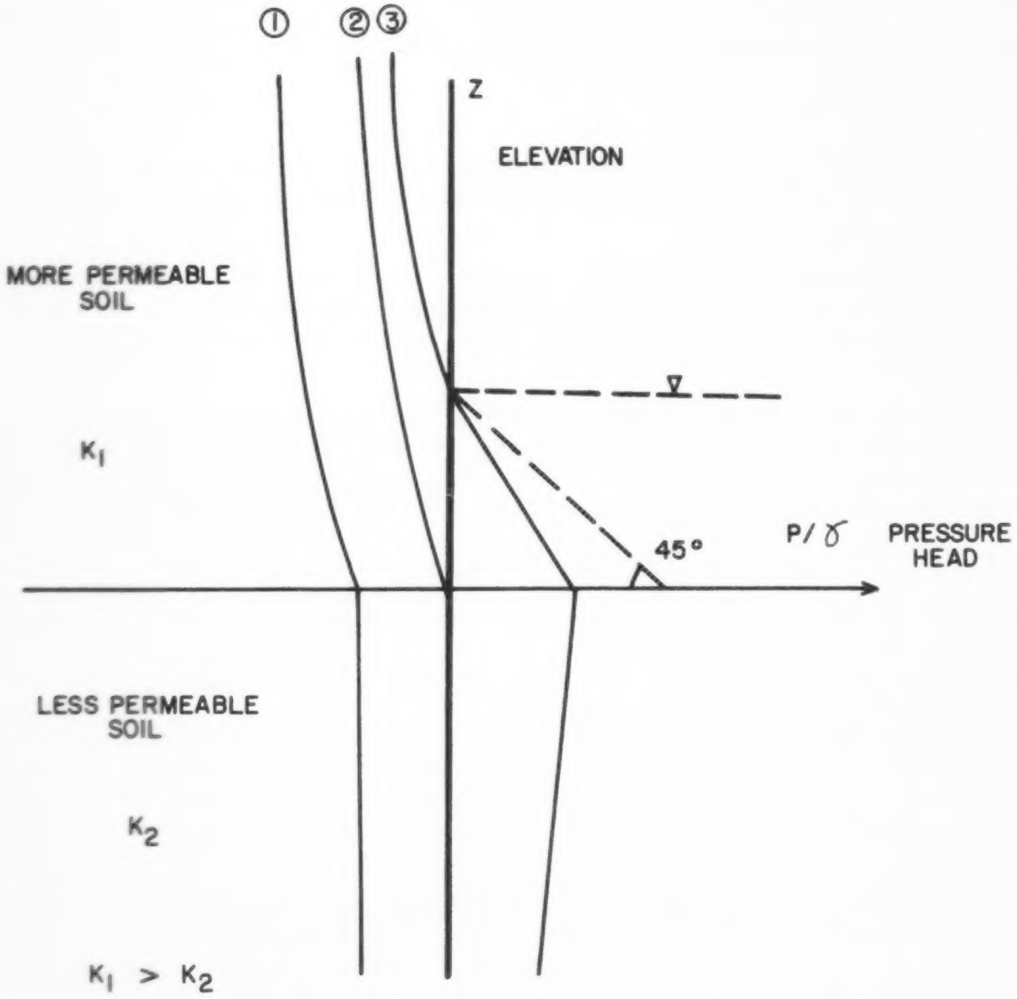

①
②
③
Z
ELEVATION
MORE PERMEABLE SOIL
K_1
45°
P/γ
PRESSURE HEAD
LESS PERMEABLE SOIL
K_2
$K_1 > K_2$

Figure 18. Pressure head diagram in a transition from a more permeable to a less permeable layer (after Bear, Zaslavsky, and Irmay, 1968). For 1, the flow rate $J < K_2$; for 2, $J = K_2$; and for 3, $J > K_2$.

TABLE 3. TYPICAL HYDRAULIC CONDUCTIVITY VALUES (after Cooley, Harsh, and Lewis, 1972).

Material	Hydraulic conductivity (gpd per square foot at 60°F)		
Granite	0.0000009	to	0.000005
Slate	0.000001	to	0.000003
Dolomite	0.00009	to	0.0002
Hematite	0.000002	to	0.009
Limestone	0.00001	to	0.002
Gneiss	0.0005	to	0.05
Basalt	0.00004	to	1
Tuff	0.0003	to	10
Sandstone	0.003	to	30
Till	0.003	to	0.5
Loess	1	to	30
Beach sand	100	to	400
Dune sand	200	to	600
Clay	0.001	to	1
Silt	1	to	10
Very fine sand	10	to	100
Fine sand	100	to	1,000
Medium sand	1,000	to	4,500
Coarse sand	4,500	to	6,500
Very coarse sand	6,500	to	8,000
Very fine gravel	8,000	to	11,000
Fine gravel	11,000	to	16,000
Medium gravel	16,000	to	22,000
Coarse gravel	22,000	to	30,000
Very coarse gravel	30,000	to	40,000
Cobbles		Over 40,000	

Although the saturated hydraulic conductivity of a soil is not dependent on the water content, K is not necessarily a constant. For example, a decrease in temperature may dissolve air bubbles entrapped in the soil pores, causing an increase in the hydraulic conductivity. The hydraulic conductivity of soils containing clays will also be affected by the cationic constituents and solute concentrations of applied water. Of particular concern with respect to the cationic composition of the applied water is the relationship of monovalent sodium to divalent calcium and magnesium. This relationship is commonly represented by the sodium adsorption ratio:

$$SAR = \frac{Na^{+}}{\sqrt{\frac{Ca^{++} + Mg^{++}}{2}}} \qquad (16)$$

where Na^{+}, Ca^{++}, and Mg^{++} are expressed in milliequivalents per liter.

SAR values greater than 6 and 9 in an irrigation source are expected to reduce the hydraulic conductivity of shrinking-swelling soils (Ayers and Westcot, 1976), thus decreasing the long-term infiltration rate. However, the ratio of carbonate and bicarbonate levels in irrigation water must be considered. As stated by Ayers and Westcot: "When drying of the soil occurs between irrigations, a part of the CO_3 and HCO_3 precipitates as $Ca\text{-}MgCO_3$, thus removing Ca and Mg from the soil water and increasing the relative proportion of Na which would increase the sodium hazard." The older concept of SAR has been adjusted to accommodate the effect of lime dissolution or deposition by means of the adjusted SAR:

$$\text{adj SAR} = \text{SAR}\left[1 + (8.4 - \text{pHc})\right] \qquad (17)$$

where Na, Ca, and Mg are obtained from water analyses. The pHc is the theoretical pH of water equilibrated with $CaCO_3$. In practice, pHc is calculated using equations given by Ayers and Westcot.

The relationship of exchangeable sodium percentage (ESP), the fraction of the exchange complex occupied by sodium, to the total salt concentration of the soil solution also affects the hydraulic conductivity of soils with clays. According to McNeal (1974), a combination of high ESP and low total salt concentration leads to swelling and dispersion of the soil minerals. Swelling reduces the hydraulic conductivity of the soil. Dispersion may also cause a migration of clay particles into lower regions of the soil profile, possibly forming a layer which restricts free water movement.

Quantitative interrelationships between ESP, total salt content of the soil solution, and hydraulic conductivity were demonstrated on laboratory soil columns by Quirk and Schofield (1955). In particular, they observed that for a given ESP, the hydraulic conductivity, K, increased with an increase in total salt content. The relationships between K, ESP, and salt content for Pachappa sandy loam (McNeal, 1974) are shown on Figure 19. This figure also illustrates that a greater and greater total salt content is required to offset the effect of sodium on K as ESP increases.

Quirk and Schofield (1955) coined the expression "threshold salt concentration" to represent the level at which the salt content would need to be reduced to produce a 10- to 15-percent reduction in K at a given value of ESP.

As pointed out by McNeal (1974), these concepts have applicability in land disposal operations as well as in irrigated agriculture, particularly if a sizable fraction of the effluent contains softened water.

Laboratory devices for measuring the hydraulic conductivity of disturbed or undisturbed soils are generally classified as permeameters. Klute (1965) describes the construction and operation of common permeameters. Soil cores taken in the field may be used directly in the permeameter. For research purposes, columns are filled with dried and sieved soils packed to the desired bulk density. Klute (1965) recommends that water to be applied to the field soil should be used in conducting the tests.

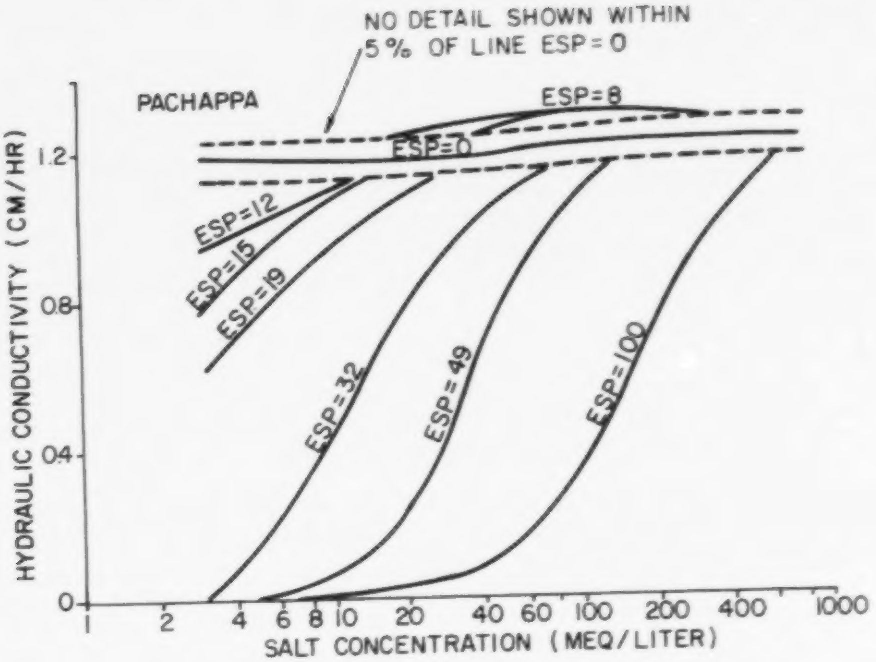


Figure 19. Relationships among hydraulic conductivity, salt concentration, and ESP for Pachappa sandy loam (after McNeal, 1974).

Two general types of permeameters that are used are the constant head type and the falling head type. Generally, the permeameters are arranged so that flow is vertically downward through the soils. For the constant head type of permeameter, water is ponded continuously at a constant distance above the surface of the soil core. The rate of water movement through the core is measured via a graduated cylinder and stopwatch. The hydraulic conductivity is calculated from Darcy's equation. In the falling head method, the soil core is connected to a standpipe. After saturating the soil, the water level in the standpipe is allowed to fall. The time for the head to drop through successive distances is measured and Darcy's equation is used to calculate K.

As Klute (1965) pointed out, the results of the permeameter methods are of questionable value when applied to a large area. Thus, a large number of samples would be required to account for the spatial variability in field hydraulic conductivity of soils. When samples are also obtained from deeper regions of the vadose zone for permeameter tests, the problem of spatial variability assumes an even more three-dimensional character.

Relationships have been developed between hydraulic conductivity and characteristic grain sizes of soils and aquifers. These relationships require conducting grain-size analyses of representative samples and employing appropriate equations. Details are presented by Freeze and Cherry (1979, pp 350-352). In general, this approach should be used with caution because of the difficulty of obtaining representative samples.

FIELD METHODS FOR DETERMINING THE RATE OF WATER MOVEMENT IN THE VADOSE ZONE

Infiltration

Recent reports by the U.S. EPA et al. (1977) and Sidle (1979) present excellent reviews of methods for estimating long-term infiltration rates. For disposal operations involving the surface spreading or ponding of wastes, the two most suitable procedures for determining long-term infiltration are cylinder infiltrometers and test basins. Infiltrometers may be either the single-ring or double-ring type. In the double-ring infiltrometer method, a metal cylinder from 6 to 14 inches in diameter is driven to a depth of 6 inches into the soil at the test site. A larger ring, ranging from 16 to 30 inches in diameter, is placed concentrically around the smaller ring. The areas within inner and outer rings are flooded and the rate of recession of water level in the inner ring is measured via a hook or point gage.

By keeping the outer region flooded, flow in the inner region is restricted mainly to the vertical direction. Wastewater to be used during the full-scale disposal operation should be used during infiltrometer tests. Because of the spatial variability of soil properties, an inordinate number of ring tests may be required to ensure that results are within a certain percentage of "true" mean values. Even with care, however, experience has shown that double-ring infiltrometers tend to overestimate the true infiltration rate because of the divergence of flow due to shallow impeding layers and unsaturated flow (Bouwer, 1978). Bouwer recommended using a single large cylinder, in lieu of a double-ring infiltrometer, to offset the problem of divergence from unsaturated flow.

A superior alternative to infiltrometers is to install a number of small test basins at the site. Test basins should be large enough to permit installation of the desired accessory facilities (see Figure 22). For studies involving an irrigated field, Nielsen, Biggar, and Erh (1973) used basins that were about 400 square feet. Basins may be constructed by digging a narrow trench around the desired rectangular area to such a depth that lateral flow will be minimized. Side boards, possibly covered with plastic sheeting, are placed in the trench, ensuring that good contact is made with the soil within the plot and also at the corners. Sufficient free-board should be allowed to permit a range of flooding heads. During tests, water is metered into the plot at a discharge rate such that a constant head is maintained. The discharge rate to maintain this head is the application rate. The infiltration rates at selected times are calculated by dividing the application rate by the surface area.

Alternatively, the head may be allowed to fall a short distance by shutting down the inflow, and the volume which infiltrates per unit area over a measured period of time may be converted to infiltration rates.

As with infiltrometers, the same water source to be used during the actual operation should be applied during the tests. The effect of such effluent on surface intake rates (e.g., by clogging) may be readily examined. Similarly, the effect of unfavorable exchangeable sodium percentages and electrolyte concentrations on the hydraulic conductivity can be quantified.

Because of the spatial variability of soil properties, even on supposedly homogeneous areas, it is recommended that a number of test basins be located on the proposed waste disposal site. For example, by arranging the basins in accordance with recommended statistical procedures (e.g., Steele and Torrie, 1960), results amenable to statistical analyses will be generated for infiltration rates and other properties of interest.

A common problem with the test basins and infiltrometers is that shallow impeding layers may promote lateral movement of water in preference to truly vertical flow. Lateral flow rates generally exceed vertical rates. Hence, measured intake rates tend to overestimate the intake rates of a larger disposal area.

If the technique for applying wastewater to the disposal site is sprinkler irrigation, it is recommended that infiltration tests be conducted using rainfall simulators. Such units are described by Bertrand (1965), Sidle (1979), and Hamon (1979).

In conducting infiltration trials, a record should be kept of the intake rate as a function of time. Plotting of the results will produce an exponential relationship between intake and time. The resultant curve will asymptotically approach a final value, specifically that of the saturated hydraulic conductivity.

Unsaturated Flow

The principal unsaturated flow parameters which require quantification at a waste disposal site following infiltration include the following: (1) the amount of water moving into the lower vadose zone, (2) the direction of unsaturated water movement, (3) hydraulic gradients, (4) the unsaturated hydraulic conductivity, and (5) flow rates (flux).

Amount of Water Movement Below the Soil Zone--

By means of an appropriate soil moisture accounting system, it may be possible to estimate the amount of water draining from the soil zone into the lower vadose zone. Such an accounting system or water balance is based on the hydrologic equation for the soil-water system. This equation is often written in the form:

$$\Delta SM = PPT - PET - RO \qquad (18)$$

where ΔSM = the change in soil moisture storage

PPT = precipitation + irrigation

PET = potential evapotranspiration

RO = runoff.

A commonly used water balance method is that developed by Thornthwaite and Mather (1957). This method, which is essentially a bookkeeping procedure, accounts for soil moisture additions and extractions on an annual basis. Basic credit components are the quantities (expressed as depths) of rainfall and irrigation. The principal losses (debits) from the system are evapotranspiration, runoff, and deep percolation. Monthly temperature and precipitation data are required. The potential water storage capacity of the soil must be estimated for the soil depth of interest. Generally, the potential storage capacity can be taken as the difference in water content (on a volumetric basis) of the soil at field capacity and that at a selected drained value.

The analyses of Thornthwaite and Mather (1957) generate values of changes in soil moisture storage. For example, if potential evapotranspiration exceeds rainfall or other water additions to the system, the depth of water in storage is decreased. In contrast, when additions exceed the potential water holding capacity, the excess either runs off or percolates. By measuring runoff, the amount of percolation is determined by differences. The most difficult component of the analysis to quantify is the consumptive use, or evapotranspiration. A recent report to the American Society of Civil Engineers (Jensen, 1973) discussed in detail the present state of the art for measuring evapotranspiration losses.

Fenn, Hanley, and DeGeare (1975) and Mather and Rodriguez (1978) used variations of the Thornthwaite-Mather method to estimate the potential of infiltrating rainwater to move into a landfill, creating leachate. Assumptions were made on the thickness and storage capacities of soil cover and solid waste cells, on packing of solid waste, and on transmission characteristics of water in solid waste. According to Mather and Rodriguez (1978), the advantages of the water budget approach are that it is easy to apply, it provides quantitative values, and it is based on straightforward assumptions. The method should find applicability to landfills in humid regions where precipitation amounts are great enough to produce surplus water in the soil layers. For landfills in arid and semiarid regions protected from sheetflow, surplus water may not be available for deep percolation into solid waste cells and the method may find limited utility.

Direction of Water Movement and Hydraulic Gradients--

Methods for monitoring water movement and hydraulic gradients in saturated regions such as perched groundwater cannot be used to monitor unsaturated flow because of the so-called outflow principle: water will not freely enter a soil cavity unless the soil-water pressure is greater than atmospheric. Consequently, special methods must be used in unsaturated soils. Of the available methods, the three most commonly used to infer unsaturated flow are tensiometers, psychrometers, and neutron moisture logging.

Tensiometers--The principles of operation, methods of installation, and limitations of tensiometers were discussed in the previous section. In characterizing the direction of unsaturated water movement, a battery of tensiometers must be installed with individual cups terminating at successive depths throughout the region of interest. Using the principles of hydraulics, the total hydraulic head, H, at a point in a flow system is the sum of the positional (gravitational) head, z, plus the pressure head, h. The positional head is the distance between the measuring point (e.g., tensiometer cup) and an arbitrary datum. That is, positional head equals zero at the datum. The pressure heads are readings taken from the tensiometer manometer system as shown on Figure 12. Positional heads are positive and pressure heads are negative during unsaturated flow. Differences in the total head between tensiometer cups may be used to estimate the direction of flow. In particular, flow occurs in the direction of decreasing total head.

Figure 20 illustrates three hypothetical flow cases in a soil profile overlying a water table. This example is an adaptation of one presented by Reeve (1965). In the example, the datum is taken at the base of the aquifer. Three tensiometers are located in the soil zone above the water table. The cups are connected to water manometers for illustrative purposes only. In a real case, the tensiometers would be connected to above-ground mercury manometers as shown on Figure 12.

In the first case, water is not moving upward or downward in the profile, as evident by calculating the total hydraulic heads. Note that the water levels in the open ends of the three manometers are equal and correspond to the water table. For unit #1, the total hydraulic head, H, is the sum of the positional head, z, and the pressure head, $-h_1$. That is, $H_1 = z_1 - h_1$. But this is simply the distance from the datum to the water table. For unit #2, $H_2 = z_2 - h_2$. Again, this equals the distance from the datum to the water table. It can be shown that the total hydraulic heat at unit #3 also equals the distance from the datum to the water table. In order for flow to occur in the soil between units, a difference in total head must exist. But $H_2 - H_1 = 0$ and $H_3 - H_2 = 0$. In other words, the system is in equilibrium. Incidentally, for such a system, measurements in tensiometers afford a means of defining the water table position.

In the second case, the elevation of the water table is different and water is flowing vertically downward in the profile. In this example, the pressure heads are all zero because the level of the meniscus is at the same level as the cups. For all cups, the positional heads remain equal to the distance from the datum to the cups. In this case, the total hydraulic head at each cup is just the value for the positional heads, z_1, z_2, and z_3. Inasmuch as the positional head (and thus total head in this case) decreases downward, flow also occurs in a downward direction.

Finally, in the third case, evaporation is occurring at the soil surface. By applying the same reasoning as above, it could be shown that the total hydraulic head decreases vertically upward in the profile so that upward movement occurs.

It is necessary to install more than one array of tensiometers to detect horizontal flow. Thus, if individual arrays terminate at varying depths, differences in total hydraulic heads between corresponding units in successively lateral arrays may suggest that horizontal movement is occurring in the unsaturated state. Because of the heterogeneity in soil properties, however, definitive conclusions on lateral flow may be tenuous.

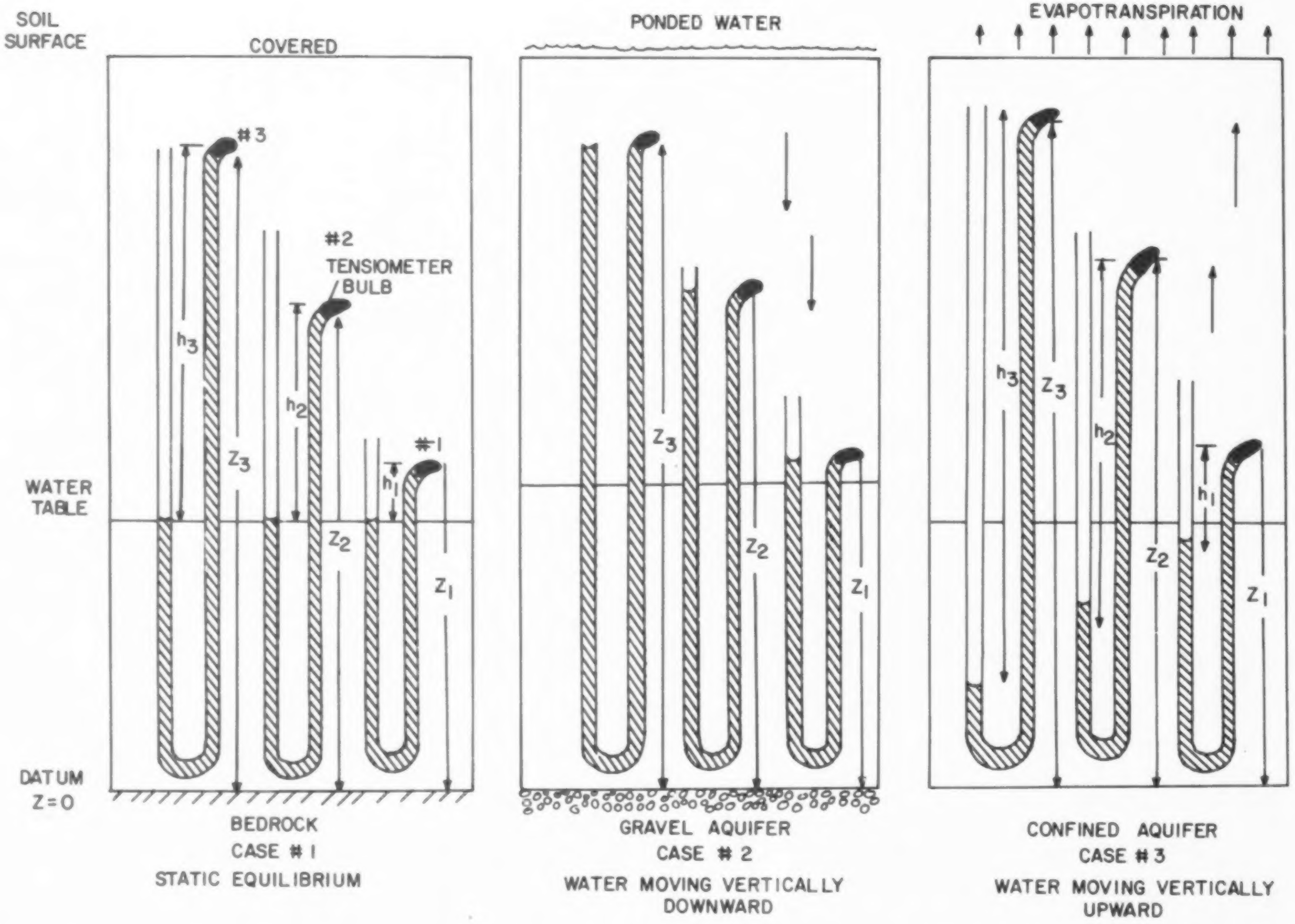

SOIL SURFACE
WATER TABLE
DATUM Z=O
COVERED
#3
#2
TENSIOMETER BULB
#1
h_3
h_2
h_1
Z_3
Z_2
Z_1
BEDROCK
CASE # 1
STATIC EQUILIBRIUM
PONDED WATER
GRAVEL AQUIFER
CASE # 2
WATER MOVING VERTICALLY DOWNWARD
EVAPOTRANSPIRATION
CONFINED AQUIFER
CASE # 3
WATER MOVING VERTICALLY UPWARD

Figure 20. Distribution of hydraulic heads for three unsaturated flow cases (after Reeve, 1965).

An array of tensiometers may be useful in detecting the presence of clogging layers and in manifesting the head loss by such layers (Bouwer, 1978). For example, in water-spreading operations, it may happen that because of dispersion, fine solids may migrate in the profile and filter out at the interface between soil materials of differing texture. A tensiometer above this layer will gradually show an increase in soil-water pressure, while a unit below the layer will shift to the more negative side.

Time lag in response to a rapidly changing system is also a problem with tensiometers. The tensiometer design of Watson (1967) involving a pressure transducer system for measuring soil-water pressure has more rapid response characteristics than mercury-water manometer systems.

Psychrometers--To measure soil-water pressure under very dry conditions, the tensiometer unit is no longer applicable because of air-entry problems. To measure lower negative pressures, therefore, other instrumentation is required. In recent years, progress has been made in developing the thermocouple psychrometer for this purpose. In-situ pressure measurements down to -100 atmospheres may be possible with these units (Warrick, personal communication, 1980).

The principle of psychrometric measurement of soil-water potential is given by Rawlins and Dalton (1967) who also diagrammed a representative unit (see Figure 21). The basic idea in soil-water psychrometry is that a relationship exists between soil-water potential and the relative humidity of soil water. This relationship is given by:

$$\Psi \equiv \frac{RT}{n} \ln \frac{P}{P_o} \tag{19}$$

where Ψ = soil water potential (matric potential plus osmotic potential)

R = ideal gas constant

T = absolute temperature

n = volume of a mole of water

P/P_O = relative humidity.

In construction, psychrometers consist of a porous bulb comprising a chamber to sample relative humidity of a soil, a sensitive thermocouple, heat sink, reference electrode, and associated electronic circuitry. Such units use the principle of Peltier cooling to lower the temperature of one junction of the thermocouple below the dewpoint, thereby allowing an evaluation to be made of the relative humidity.

Earlier units were plagued with problems relating to temperature effects on measurement of potential. By designing the instruments to meet certain boundary conditions, however, these problems were overcome (Merrill and Rawlins, 1972).

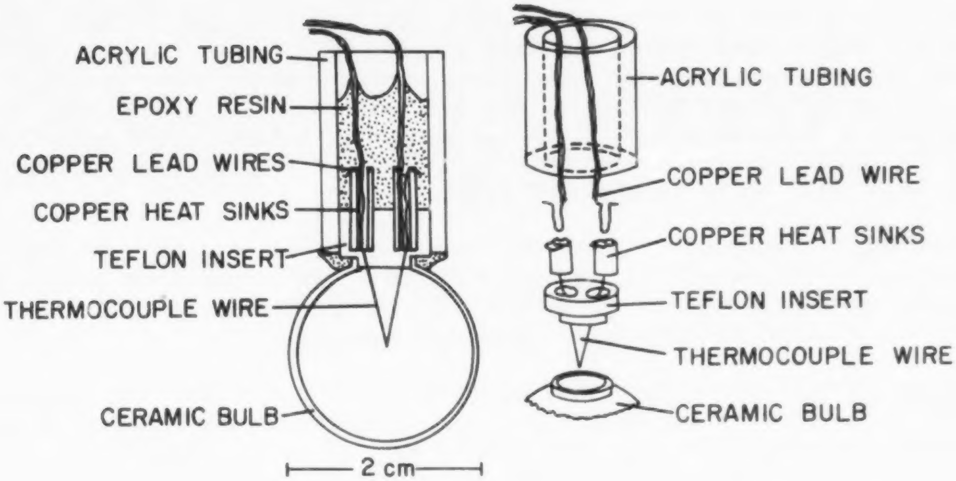

ACRYLIC TUBING
EPOXY RESIN
COPPER LEAD WIRES
COPPER HEAT SINKS
TEFLON INSERT
THERMOCOUPLE WIRE
CERAMIC BULB
2 cm
ACRYLIC TUBING
COPPER LEAD WIRE
COPPER HEAT SINKS
TEFLON INSERT
THERMOCOUPLE WIRE
CERAMIC BULB

Figure 21. Soil psychrometer (after Rawlins and Dalton, 1967).

Operation of the Peltier-effect psychrometer consists of (Merrill and Rawlins, 1972):

> (1) measuring the electromotive force (e.m.f.) while the measuring junction is dry (dry e.m.f.); (2) passing a current through the sensor, cooling the measuring junction below the dewpoint, and thereby wetting it; (3) measuring the e.m.f. again after an initial, transient phase of the e.m.f. has subsided, but before the e.m.f. has decayed significantly as a result of depletion of water on the junction (wet e.m.f.); and (4) again passing current through the thermocouple wires in the reverse direction to head and dry the measuring junction. This mode of operation permits any temperature difference between the measuring junction and the reference junctions to be compensated for by subtraction of the dry e.m.f. from the wet e.m.f. (Δ e.m.f.).

Two types of thermocouple psychrometers are available. One unit, installed in access tubes, consists of positioning psychrometers in porous cups at the base of tubing. This unit may be withdrawn for recalibration. The second type of unit, called "sealed-cup" psychrometers by Merrill and Rawlins (1972), contains the thermocouple unit permanently sealed into a porous enclosure.

Each unit must be calibrated prior to field installation. Techniques for such calibration are given by Merrill and Rawlins. Also in field installations, it is important to account and correct for diurnal changes in temperature. Installation of a sensitive thermistor is recommended. Merrill and Rawlins point out that field psychrometers could be tied into a remote sensing scheme, provided care is exercised in instrumentation.

Enfield, Hsieh, and Warrick (1973) used thermocouple psychrometers to estimate the direction and rate of water flux in material in a vadose zone with a water table at 310 feet in a desert environment in Washington. Individual psychrometric units were placed 3 meters apart within a specially designed well casing. In addition to the thermocouple psychrometers for measuring the matric potential, temperature transducers were also installed, providing data to relate moisture flux due to temperature gradients.

Neutron moisture logging--Although the neutron thermalization technique is used mainly to determine changes in volumetric water content of materials in the vadose zone, the method may also be used to infer water movement. In particular, if a soil-water characteristic curve is available for incremental depths throughout the vadose zone, it may be possible to relate water content values to water pressure. Hydraulic head gradients, and therefore flow direction, could then be inferred. Unfortunately, the accuracy of the method may not be great enough to detect slight water content changes, particularly in the dry range.

Wilson and DeCook (1968) and Wilson (1971) used moisture logs from a network of 100-foot deep access wells to infer the rates of lateral movement of recharge waves in the vadose zone during river recharge and during artificial recharge. The arrival of such waves was inferred by the change in water content in a perched mound at about 33 feet.

Flux, Velocity, and Unsaturated Hydraulic Conductivity--

As discussed earlier, one of the prime goals of vadose zone monitoring is to estimate the flux and velocity of water in this zone. Some of the methods for estimating flux and velocity are based directly on the equation of continuity and information on hydraulic conductivity values is not required. However, other methods are based on Darcy's equation and values of both hydraulic conductivity and hydraulic gradients are necessary. In this report, both classes of methods are referred to as "draining-profile" techniques because measurements are made during drainage cycles. Other methods reviewed are: (1) flow meters for direct measurement of flux, (2) use of suction cups for estimating velocity, (3) tracer techniques, and (4) an approximate method using water budget data.

It should be emphasized that, by and large, the methods presented may be suitable for characterizing flux and hydraulic conductivity in soils and may have limited applicability to deeper regions of the vadose zone. However, it is important in many waste disposal operations to determine loss in the near-surface region. For example, during site evaluation of hazardous waste disposal ponds, a series of tests could be conducted to determine a site with minimal K and J values.

Draining-profile methods--The simplest approach for estimating flux at successive depths in the vadose zone is to obtain a series of water content profiles during drainage. The method is essentially based on the equation of continuity, expressed in terms of flux:

$$\frac{\partial\theta}{\partial t} = \frac{\partial J}{\partial z} \qquad (20)$$

or

$$J = -\int_0^z \frac{\partial\theta}{\partial t}\, dz \qquad (21)$$

To account for all sources and sinks, Bouwer and Jackson (1974) suggested using the following expression for flux:

$$J_1 = R + I - ET - \int_0^z \left(\frac{\partial\theta}{\partial t}\right) dz \qquad (22)$$

where J_1 = combined flux

R = rainfall

I = irrigation

ET = evapotranspiration.

For a draining profile covered to prevent ET, equation 22 reduces to equation 21.

An example of hypothetical water content profiles from a field site is shown on Figure 22 (after Bouwer and Jackson, 1974). Calculation of flux out of a given depth, z, between successive times, t_1 and t_2, requires the solution of equation 21. Inasmuch as the right-hand side of the equation is simply the shaded area of the sketch divided by the time difference $t_2 - t_1$, the flux may be found by graphical integration of water content profile between successive measurement dates.

Water content profiles could be obtained using techniques discussed in another section, e.g., neutron moisture logging or tensiometers. An advantage of neutron moisture logging is that profiles can be obtained at considerable depth, provided access wells are installed carefully to prevent side leakage. The sequence to obtain water content profiles using tensiometers is as follows:

1. Install tensiometers at sequential depth intervals through the soil region of interest.

2. Obtain soil cores from the soil at depths bracketing the tensiometer cups.

3. Obtain water characteristic curves for each core in order to define the relationship between water content and soil-water pressures. Inasmuch as

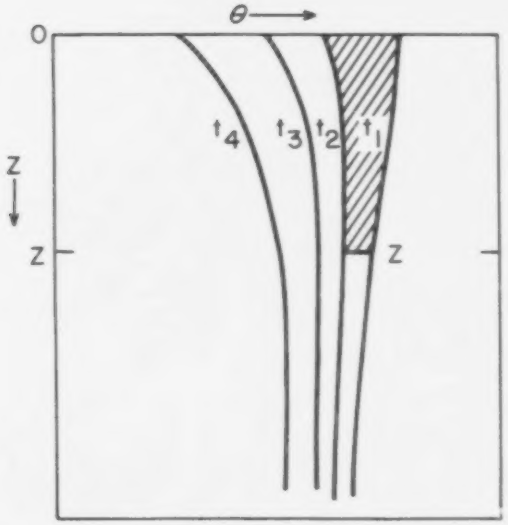


Figure 22. Hypothetical water content profiles at several times during drainage of a soil profile (after Bouwer and Jackson, 1974).

field data will be obtained during drainage, curves must be obtained for drying cycles.

4. Determine pressure values in tensiometers during drainage of the soil profile. See Figure 23(a).

5. Convert pressure readings to water content values using the appropriate water characteristic curves. See Figure 23(b).

6. Plot water content versus soil depth. See Figure 23(c).

An alternative draining-profile method requires two stages: (1) determining the hydraulic properties of soil in a number of plots within a field or disposal area, and (2) using the resultant hydraulic properties and appropriate instrumentation to estimate flux at additional field locations. Basically, the method is an adaptation of a two-step technique presented by LaRue, Nielsen, and Hagan (1968) for estimating soil-water flux below the root zone of an irrigated field.

During step 1, a number of test plots are established at random locations in the area of interest. Each plot is instrumented with a depth-wise sequence of tensiometers (installed in triplicate at each depth) and possibly an access tube to permit moisture logging. These units should be installed in the center of the plot to minimize the effects of lateral flow. The periphery of each plot is provided with side walls to facilitate applying a head of water at the soil surface. Figure 24 shows a possible arrangement for a test plot. Soil cores obtained during installation of tensiometers are used to construct water characteristic curves for distinct depth intervals in the profile. The following procedure is used:

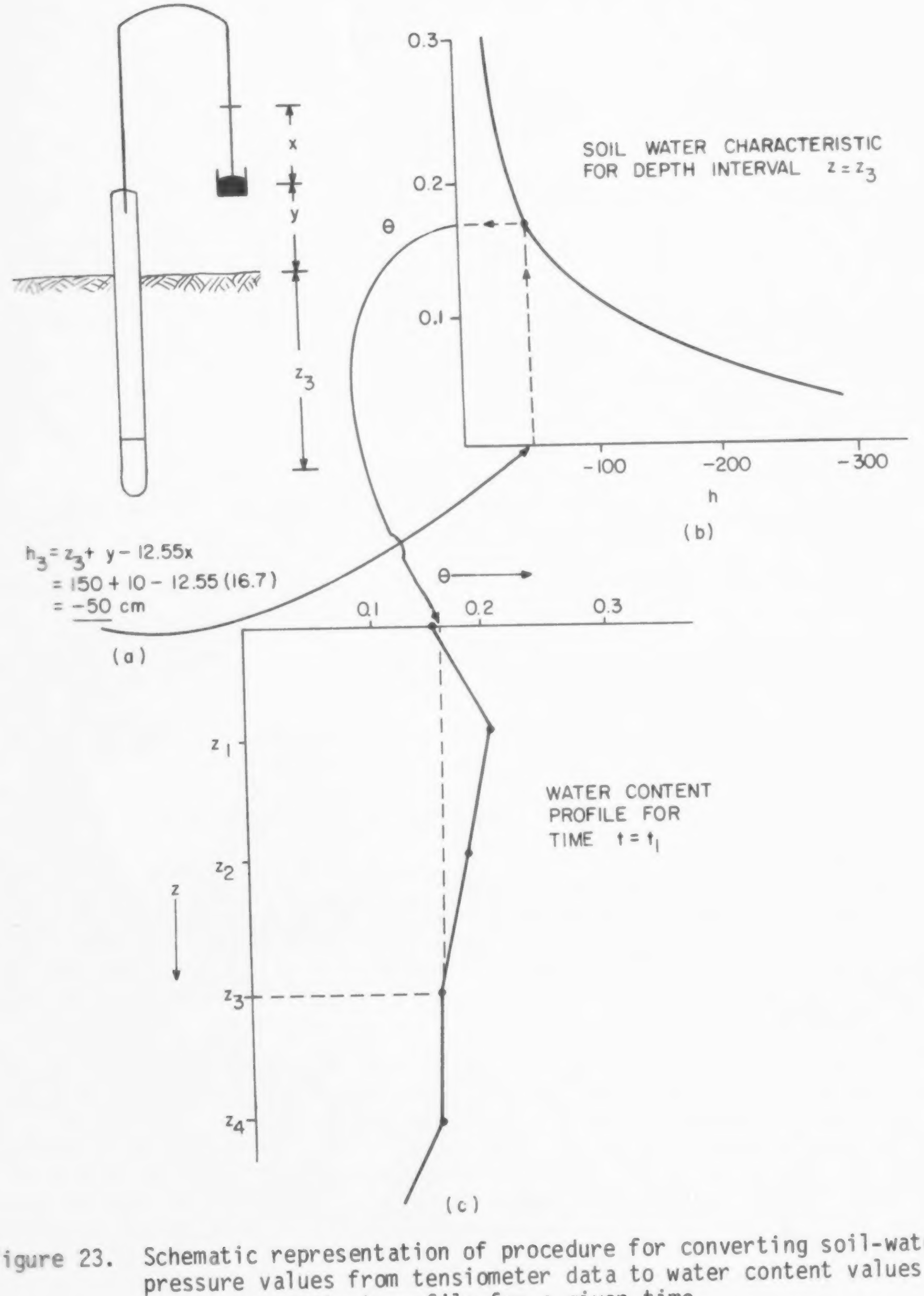

x
y
z_3
0.3
0.2
0.1
θ
SOIL WATER CHARACTERISTIC FOR DEPTH INTERVAL $z = z_3$
-100
-200
-300
h
(b)
$h_3 = z_3 + y - 12.55x$
$= 150 + 10 - 12.55\,(16.7)$
$= \underline{-50}$ cm
(a)
θ ⟶
0.1
0.2
0.3
z_1
z_2
z_3
z_4
z
WATER CONTENT PROFILE FOR TIME $t = t_1$
(c)

Figure 23. Schematic representation of procedure for converting soil-water pressure values from tensiometer data to water content values on on a water content profile for a given time.

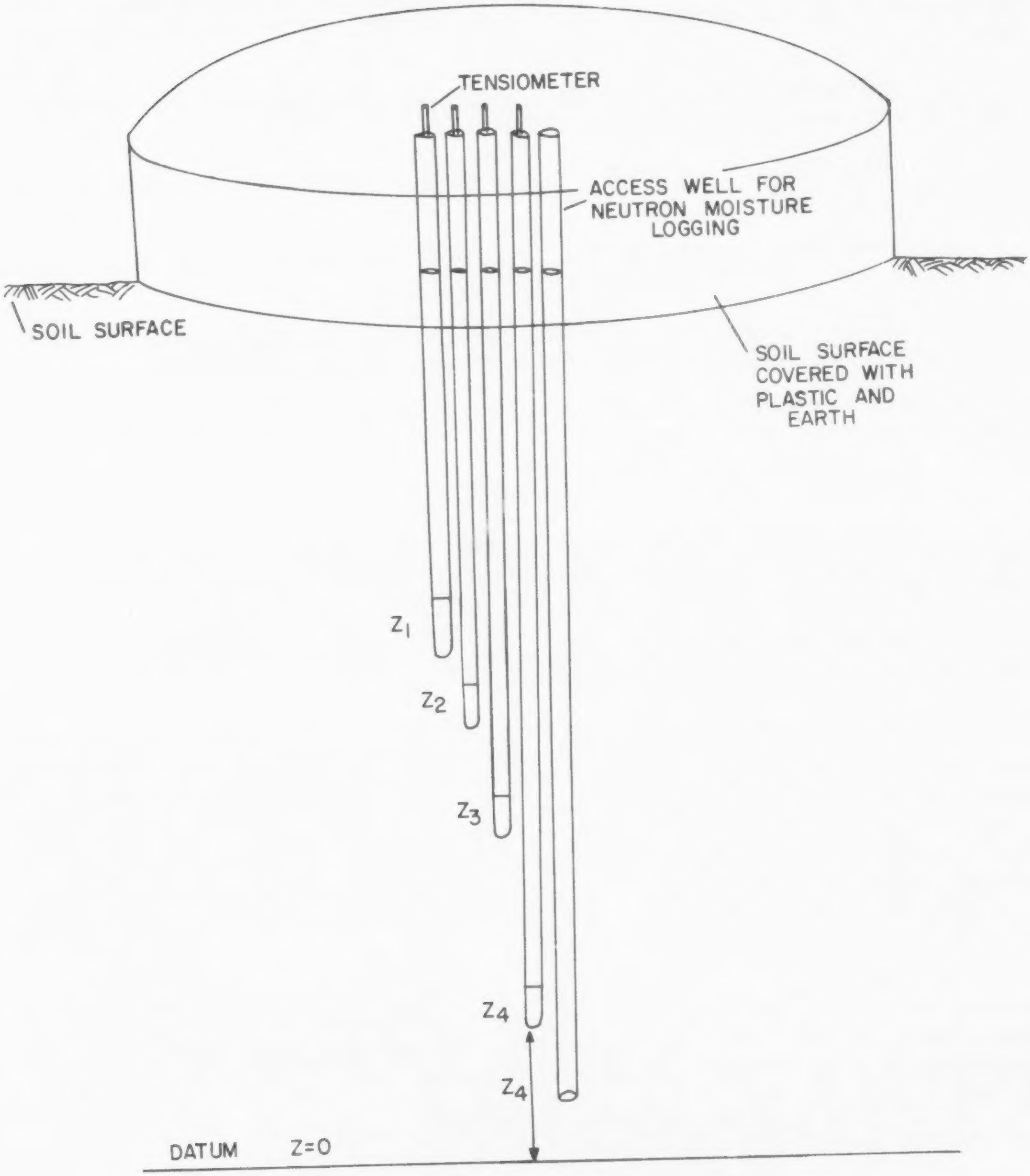

TENSIOMETER
ACCESS WELL FOR NEUTRON MOISTURE LOGGING
SOIL SURFACE
SOIL SURFACE COVERED WITH PLASTIC AND EARTH
Z_1
Z_2
Z_3
Z_4
Z_4
DATUM
Z=0

Figure 24. Sketch of field plot for determining K and J showing tensiometers and access well.

1. Flood each plot until the reading on the lowermost tensiometer indicates that a maximum degree of saturation has been reached in the surrounding soil.

2. Following disappearance of free water, cover the soil surface with plastic and earth to prevent evaporation.

3. Allow the soil to drain, taking regular tensiometer readings and logging the access well.

4. From water content profile, estimate flux out of various depth intervals using the graphical approach discussed above.

5. Calculate hydraulic gradients for the selected depth intervals for each sequence of tensiometer readings averaged over the time interval.

6. Use the hydraulic gradients and flux values to calculate hydraulic conductivity from Darcy's equation,

$$J = -K(\theta)\nabla H.$$

7. Plot the $K(\theta)$ values versus average water content values (θ) for each depth interval. Note that only one $K(\theta)$ value is obtained for each time interval.

8. Repeat steps 4 through 7 for additional time steps and obtain K versus θ curves for each depth interval.

An example (courtesy, A.W. Warrick) will clarify the above procedure for the first step of the method of LaRue, Nielsen, and Hagan. Assume that a test plot has been established with tensiometers installed for measuring water pressures at discrete depths to 200 cm. An access well has been installed for obtaining water content profiles by a neutron moisture meter. The soil profile is wetted to 200 cm, following which the surface is covered and water content values and soil-water pressures are measured. Resultant water content profiles for 24 hours and 40 hours after the beginning of drainage are shown on Figure 25.

The area between the curves (shown by cross-hatching) was found by planimetering to be 2 cm (0.79 inch) of water. Pressure head values measured on tensiometers at the 120-cm (47.2-inch) and 150-cm (59.1-inch) depths after 24 hours and 40 hours of drainage were as follows:

	h_1(120 cm; 47.2 in)	h_2(150 cm; 59.1 in)
t = 24 hours	-36 cm (-14.2 in)	-29 cm (-11.4 in)
t = 40 hours	-40 cm (-15.7 in)	-32 cm (-12.6 in).

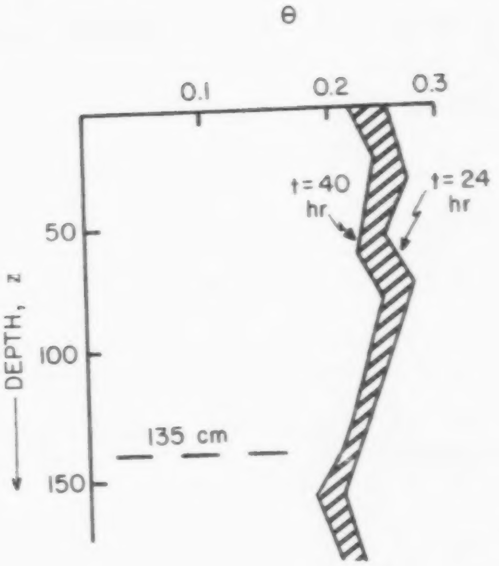


Figure 25. Water content profiles in hypothetical soil column at times t = 24 hours and t = 40 hours.

The problem is to find a K versus θ relationship for the 120-cm (47.2-inch) to 150-cm (59.1-inch) depth interval. Note that only one value can be found for one time interval; for additional values, profiles for other time intervals would need to be given. The procedure is as follows:

1. Find the drainage for the time interval t = 24 hours to t = 40 hours. Assume that we are precisely interested in the 135-cm (53.1-inch) depth (half-way between 120 cm (47.2 inches) and 150 cm (59.1 inches)). Inasmuch as the area on Figure 25 corresponds to 2 cm (0.79 inch) and the time interval is 40 - 24 = 16 hours, then the Darcian flux is:

$J = 0.125$ cm/hr.

2. Next find the hydraulic gradient. If the reference is taken at the level of the lowermost tensiometer and the positive direction is upward, the hydraulic heads at 120 cm (47.2 inches) and 150 cm (59.1 inches) at 24 hours are:

$H_{150} = (0 - 29) = -29$ cm (-11.4 inches)

$H_{120} = (30 - 36) = -6$ cm (-2.4 inches).

The gradient at 24 hours is:

$\nabla H = [-29 - (-6)]/30 = -0.76$ (24 hours).

Similarly, the hydraulic heads at 120 cm (42.2 inches) and 150 cm (59.1 inches) at 40 hours are:

$$H_{150} = (0 - 32) = -32 \text{ cm (12.6 inches)}$$

$$H_{120} = (30 - 40) = -10 \text{ cm (3.9 inches)}.$$

The hydraulic gradient at 40 hours is:

$$\nabla H = [-32 - (-10)]/30 = -0.73 \text{ (40 hours)}.$$

The above values represent averages over the 120- to 150-cm (47.2- to 59.1-inch) depth interval or, alternatively, at the 135-cm (53.1-inch) depth.

3. Calculate the hydraulic conductivity from Darcy's law. Using J from step 1 and the hydraulic gradient at 135 cm (53.1 inches) averaged over the period 24 to 40 hours:

$$J = -K(\theta)\nabla\overline{H}$$

where $\overline{H} = \frac{-0.76 - 0.73}{2} = -0.75$,

$$0.125 = -K(\theta)\ (-0.75)$$

or $K(\theta) = 0.167$ cm/hr (0.007 in/hr).

This $K(\theta)$ value corresponds to an average water content value (0.23 cm^3/cm^3) in the interval 120 cm (47.2 inches) to 150 cm (59.1 inches) during the period 24 to 40 hours. The $K(\theta)$ and θ values thus provide one point on a $K(\theta)$ versus θ curve for this depth interval. [Alternatively, K corresponds to an average h value for the four input values; $h_{avg} = (-36 - 40 - 29 - 32)/4 = -34.2$ cm (13.5 inches). The corresponding θ value could be found from a water characteristic curve for the depth interval.]

When the hydraulic properties of the field have been estimated, the procedure to follow at additional field sites during step 2 of the LaRue, Nielsen, and Hagan methods is as follows:

1. Install tensiometers at depth intervals of interest, corresponding to intervals used on the test plots. Alternatively, to minimize expense, two tensiometers could be used to bracket the lowermost interval.

2. Measure water pressures during drainage cycles using tensiometers. Calculate hydraulic gradients.

3. From the water characteristic curve corresponding to the depth interval of interest, convert water pressure value to an average θ value.

4. Using the K versus θ curve for the interval, convert θ values to K values.

5. Use the K and hydraulic gradient values with Darcy's equation to calculate flux.

Nielsen, Biggar, and Erh (1973) evaluated a number of simplified techniques for measuring flux and hydraulic conductivity values in the field. One of the promising methods is based on the following relationship:

$$J_L = K_o\ (1 + a\ K_o tL^{-1})^{-1} \qquad (23)$$

where K_o = steady-state hydraulic conductivity

a = an empirical factor relating K and θ

J_L = flux at depth L

t = time of observation.

According to Warrick and Amoozegar-Fard (1980), this equation is based on the following assumptions:

1. The soil is initially at a uniform water content θ_o
2. The unsaturated conductivity is given by the equation:

   $$K = K_o \exp\left[a(\theta - \theta_o)\right] \qquad (24)$$

3. A unit hydraulic gradient exists at all times
4. Evaporation and transpiration are negligible.

With the hydraulic gradient equal to unity, the equation of continuity, expressed by equation 13, may be written in the following form:

$$\int_0^L \frac{\partial\theta}{\partial t}\, dz = K(\theta)|_{z=L} \qquad (25)$$

or

$$L\,\frac{d\bar{\theta}}{dt} = K(\theta)|_{z=L} \qquad (26)$$

where $\bar{\theta}$ is average soil-water content in profile above a level L. The hydraulic conductivity is evaluated at depth z = L.

In practice, hydraulic conductivity values are determined from test plots (e.g., Figure 24) using the following procedure: (1) measuring θ via a neutron moisture meter or (2) converting pressure readings from tensiometers to θ values using appropriate water characteristic curves, and (3) solving equation 26 for $K(\theta)$ at depths of interest. If a graph of ln $K(\theta)$ versus

θ produces a straight line, equation 24 is assumed to be valid and we may proceed to use equation 23 to calculate J_L. Incidentally, the graph is also used to determine the factor a, which is simply the slope of the straight line.

Nielsen and Biggar (1973) presented a plot of ln $K(\theta)$ versus θ for test plot data for Yolo loam, an unusually uniform soil. The scatter of points was minimal. For a more heterogeneous soil profile, values of $K(\theta)$ ranged over four orders of magnitude for a given water content. However, the flux of water at a given depth was still adequately described using an average straight line from the ln $K(\theta)$ versus θ plot. Problems arising from the spatial variability in soil properties are described in a later section.

It should be noted that equation 26 provides us with yet another alternative method for estimating flux. That is, the left-hand side of the equation is an alternative form for expressing the flux. Thus:

$$L \frac{d\overline{\theta}}{dt} = J_L = K(\theta)\big|_{z=L}$$

In other words, if the hydraulic gradient is equal to unity in the flow system, the flux at a depth equals the corresponding hydraulic conductivity value.

Flux values determined from the draining-profile techniques are converted to values of average linear velocity by dividing by the appropriate water content values. For example, if the calculated flux of a soil is 0.125 cm/hr and the average water content is 0.23, the average linear velocity is:

$$\begin{aligned} \overline{v} &= J/\theta \qquad (27) \\ &= 0.125/0.23 \text{ cm/hr} \\ &= 0.54 \text{ cm/hr.} \end{aligned}$$

Direct Measurement of Flux

Attempts have been made in recent years to develop equipment to measure soil-water flux in the unsaturated state which does not require information on the hydraulic conductivity.

Two type of flow meters reported by Cary (1973) involve (1) direct flow measurement, and (2) the displacement of a thermal field by water in motion. The direct flow unit involves measuring the flow of soil water intercepted by a porous tube containing a sensitive flow transducer. The second unit entails measuring very accurately the transfer of a heat pulse in water moving with a porous cup buried in the soil. Because of the intimate contact between the soil and porous cup, the water moving in the cup forms a continuum with soil water. Laboratory calibration curves are prepared to relate output of a sensitive millivolt recorder during imposition of heat pulses to empirically measured flow rates.

For installation into field soils, the porous discs containing either flow transducers or heat sources are mounted in cylinders which are buried in the soil. A limitation, therefore, is that flow is measured in disturbed soils.

According to Dirksen (1974a), the accuracy of the above flow meters can be improved by:

1. Minimizing convergence or divergence
2. Extending the range of water fluxes and soil types
3. Reducing or preferably eliminating very tedious calibration procedures
4. Minimizing disturbance of soil during installation and its effect on original flow pattern.

Dirksen (1974b) discussed the design of a flux meter he developed. A feature of his unit which circumvents some of the difficulties above is that, as soil hydraulic conductivity changes, the resistance of the meter is adjusted so that the head loss across the meter matches the head loss in the soil as measured by nearby tensiometers. The fluxes through the meter and soil then are equal. Dirksen also presented a method for installing his flux meter into soils to effect a minimum of disturbance.

To date, no data are available on the use of such flow meters in the lower vadose zone of deep profiles.

Indirect Estimates of Velocity Using Suction Cups

An indirect estimate of vertical velocity in the vadose zone beneath a surface source entails observing the response of suction cups as the wetting front moves into the profile. These devices are described in detail in a later section of this report. Basically, the technique involves periodically attempting to sample from an array of suction samplers within the vadose zone. Generally, when the surrounding pore-water system is unsaturated, very little, if any, sample will be obtained. However, when the wetting front reaches a particular cup, samples are more readily obtainable. By observing the response of depth-wise units, an apparent vertical velocity may be inferred.

Meyer (personal communication, 1978) used this method to follow the wetting front during deep percolation of irrigation water in the San Joaquin Valley. Signor (personal communication, 1979) used a similar approach during recharge studies in Texas.

Tracers

A direct method for measuring the average linear velocity of water movement in the vadose zone entails determining the depth-wise arrival times of tracers introduced at the soil surface. A number of alternative tracers are available including physical tracers such as temperature, ionic constituents such as chloride, organic tracers such as fluorescent dyes and fluorocarbons, and radioactive tracers such as tritium. Evans, Sammis, and Warrick (1976) evaluated the relative merits of two tracers, temperature and tritium, for determining flux beneath an irrigated field. Results were compared with flux values found by using Darcy's equation. In general, it was found that results using temperature and tritium compared poorly with calculated values. They pointed out that problems with tritium include a short half-life (12.26 years) and possible reactions with clay particles.

Frissel et al. (1974) compared ^{36}Cl, ^{60}Co, and tritium for tracing moisture movement in laboratory soil columns. From observed breakthrough curves for the tracers, it was determined that for clay and sand soils, tritium adsorption was negligible but that the use of ^{36}Cl and ^{60}Co ions would lead to errors because of anion exclusion.

A basic problem with fluorescent tracers is sorption on clays resulting in errors in estimating flow velocities. Fluorocarbons may have promise, although sorption occurs on organic constituents of a soil (Brown, personal communication, 1979).

Simplified Method for Estimating Velocity Using Water Budget Data

For many disposal operations, it may not be possible to use the above refined methods for estimating the velocity of water-borne pollutants in the vadose zone. An approximate procedure is required to estimate whether pollutants have reached the water table. The following is suggested as one such approach. The method is based on a variation of equation 1, expressed as:

$$d_{vz} = \frac{d_w}{\theta} \qquad (28)$$

where d_{vz} = depth of penetration of water below a given soil depth

d_w = depth of water applied

θ = volumetric water content at field capacity.

The method assumes vertical piston movement of water beneath the disposal site with water content values equal to field capacity.

In practice, an estimate of seepage rate is determined using a water budget method such as that of Thornthwaite and Mather (1957), or other simplified methods. The rate is converted into an equivalent depth of water per unit time. Average field capacity values for materials in the vadose zone are estimated from drillers' logs.

As an example, assume that the depth of water seeping beneath a disposal facility for 1 year is 5 feet. If the volumetric water content corresponding to field capacity is 15 percent, the depth of penetration in the vadose zone is:

$$d_{vz} = \frac{5.0}{0.15}$$

$$= 33 \text{ feet.}$$

Thus, the rate of penetration of 5 feet of seepage is 33 feet per year. If the vadose zone is 100-feet thick, it would require about 3 years for water to reach the water table.

Effect of Spatial Variability on Soil Properties

When using field data to estimate such properties as flux and hydraulic conductivity, it is important to keep in mind that variations in these parameters may exist because of soil heterogeneity. For example, large variations in soil texture and structure, even with a given mapping unit, will affect values of related properties such as water content, porosity, and hydraulic conductivity.

The problem of such spatial variabilities was examined in detail by Nielsen and Biggar (1973), Coelho (1974), and Guma'a (1978). For their study area, Nielsen, Biggar, and Erh (1973) found that variations in water content were normally distributed with depth and horizontal distance in the field, whereas hydraulic conductivity values were log-normally distributed. The following conclusion is particularly noteworthy:

> Even seemingly uniform land areas manifest large variations in hydraulic conductivity values. Variations in texture, bulk density, and water content are much less. For a given location, methods for measuring water content, hydraulic conductivity, and hydraulic gradients will yield values that are much more accurate than required to characterize an entire field because of the heterogeneity of the soil. Thus, our ability to make predictions over a large area from a single plot can range from good to unsatisfactory, depending on the particular prediction parameter of interest.

Based primarily on the results of studies by Nielsen, Biggar, and Erh (1973) and Guma'a (1978), the variablilities in various soil parameters were ranked as follows by Warrick and Amoozegar-Fard (1980):

Low variability: (Coefficient of variability less than 20 percent)
- Bulk density
- Water content at a zero tension

Medium variability: (Coefficient of variation 20 to 75 percent)
- Textures (sand, silt, or clay)
- Field water content
- Water content at specified tension between 0.1 and 15 bars

High variability: (Coefficient of variation greater than 100 percent)

- Saturated hydraulic conductivity
- Unsaturated hydraulic conductivity
- Apparent diffusion coefficient
- Pore water velocity

Problems arising from the spatial variability of hydraulic properties of the vadose zone will increase as we proceed deeper in the profile. For example, even small lenses may cause an effect on soil-water pressures, hydraulic conductivity, and flux. Consequently, it may be that the best we can expect is to estimate flux out of the soil zone of a waste disposal facility.

Saturated Flow in the Vadose Zone

Saturated flow occurs in the vadose zone primarily within perched groundwater bodies which develop at the interface of regions of varying hydraulic conductivity (see Figure 18). The principal saturated flow characteristics which should be quantified for the vadose zone are the following: (1) hydraulic gradients, (2) direction of flow, (3) hydraulic conductivity values, and (4) flow rates.

Hydraulic Gradients and Flow Direction in Perched Groundwater--

Two basic tools for measuring hydraulic gradients in perched groundwater are the piezometer and observation well. As will be shown later, these devices are also useful for obtaining water samples for analyses.

Piezometers--Piezometers consist of small-diameter pipes which are driven, augered, jetted, or otherwise drilled into a saturated zone or a zone in which saturation is expected. Reeve (1965) discussed in detail common techniques for installing and cleaning new piezometers. In general, it is essential that a tight fit be maintained between the outer wall of the piezometer and the surrounding media. For shallow units, piezometers may be installed by augering and driving with a sledge hammer. Deeper units will require jetting or use of standard drilling equipment. It may be necessary to fill the cavity between wells and boreholes with grout to ensure tightness of fit. As with regular wells, piezometers should be developed by pumping or bailing to clean and open up the material at the base of the unit. In some cases, it may be necessary to install piezometers with screened well points to prevent the upward movement of saturated sediment into the unit.

In many field situations, it is desirable to install a number of piezometers at successive depths within a perched groundwater body. Two alternative methods of construction are possible: (1) installing a battery of separate piezometer wells, or (2) installing a cluster of wells within a common borehole. Clustered wells are constructed by installing the individual units in the borehole at desired depths and surrounding the well points with sand. The region of the borehole between units is backfilled with bentonite. Fenn et al. (1977) presented additional details on methods for installing piezometer clusters together with their advantages and disadvantages.

In operation, the piezometer provides a measure of the hydraulic head at the terminus of the tube. The total head consists of the sum of gravity and pressure heads. The gravity head is referenced to an arbitrary datum, say, the ground surface. For a nest of piezometers, it is convenient to take the elevation of the lowermost unit as the datum. The pressure head is the height of water above the bottom opening of the unit.

Depth to water in piezometer units is measured by chalked tape, electric sounders, or air lines. By referencing piezometric values to a common datum, mapping of the piezometric surface will be facilitated. Such mapping will provide clues on the direction of water movement.

Although piezometers are generally used in the soil zone, they may also be installed into the lower vadose zone. For example, Wilson and DeCook (1968) describe units which were placed into perched groundwater within the 80-foot thick vadose zone of a recharge site near Tucson, Arizona. The depths of individual units were based on moisture logs in a network of access wells. Each piezometer contained a screened well point, and grout was poured between the outside of the casing and drill hole to inhibit side leakage.

An array of piezometers with the ends of individual units terminating at different depths may be used to monitor the vertical movement of water in perched water tables. As an example (adapted from Reeve, 1965), Figure 26 indicates three hypothetical flow situations in a soil containing three piezometer units beneath a water table. For purposes of calculation, the datum is selected at the base of the lowermost piezometer. For the first case, that of perched groundwater above an impermeable layer, vertical flow is not occurring. This may be verified by calculation. The total hydraulic head for the water levels in all wells is the same and corresponds with the water table. It will be recalled that water flows in the direction of decreasing total hydraulic head. For unit #1, the gravitational head, z_1, is zero and the pressure head is h_1. Similarly, for unit #2, the positional head is z_2 and the pressure head is h_2. But $h_2 + z_2 = h_1$. Consequently, $H_1 = H_2$ and no flow occurs between units #1 and #2. Similar analysis would show that flow does not occur between units #2 and #3.

For the second case, water is applied to the surface causing water movement downward in the profiles. The perched water body is underlain by an unconfined gravel aquifer. In this case, flow occurs in the vertically downward direction. Water levels in the individual piezometers are at the base of the unit; that is, pressure heads are all zero. For unit #1, the total hydraulic head is zero because $z_1 = 0$ and $h_1 = 0$. For unit #2, the positional head equals z_2 and the pressure head = 0. The total hydraulic head at unit #2 is z_2. Inasmuch as $H_2 > H_1$, flow occurs between units #2 and #1. Also, because the vertical distance between the lower ends of units #1 and #2 is z_2, the hydraulic gradient is unity, i.e.:

$$\frac{z_2 - 0}{z_2} = 1 .$$

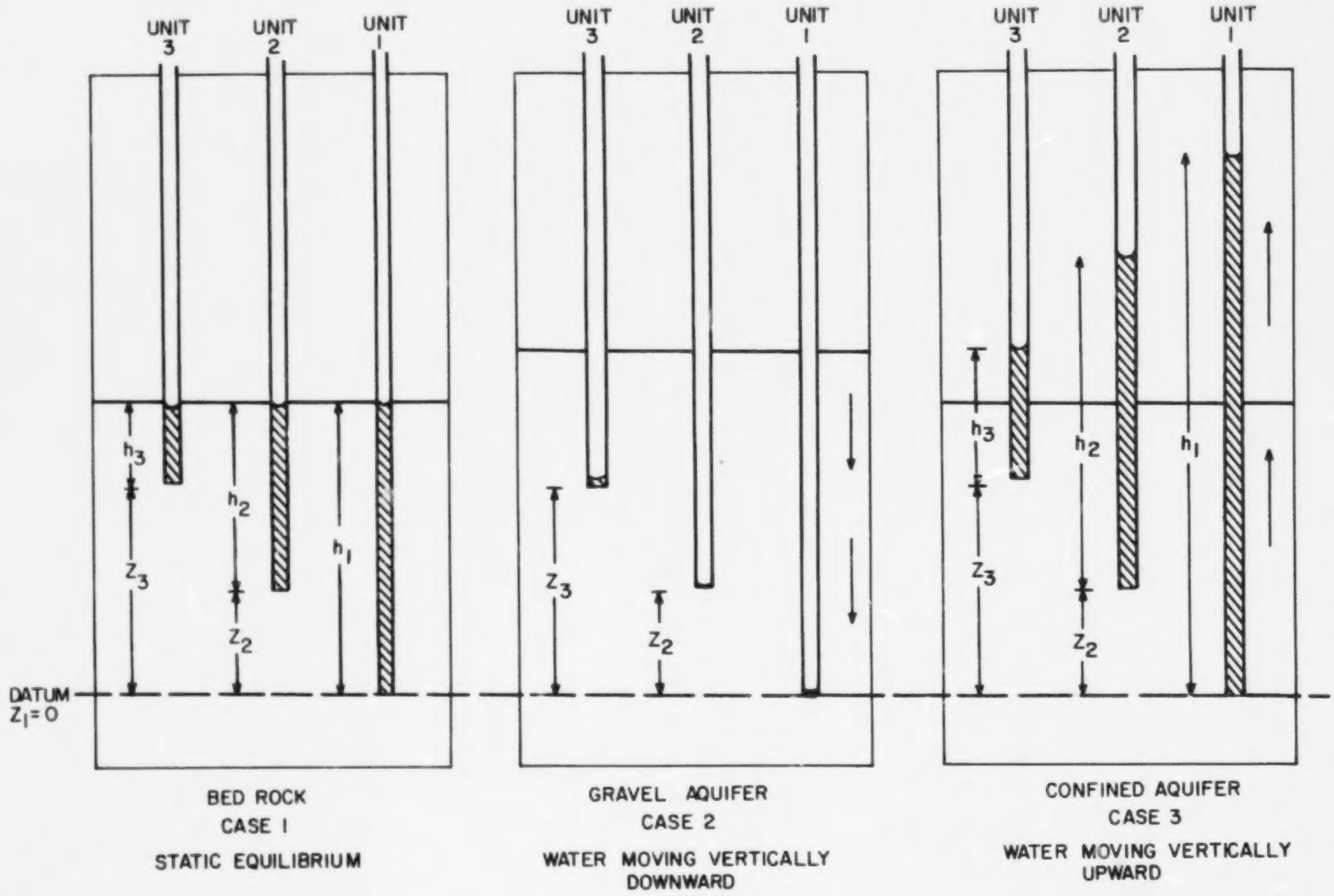

UNIT 3
UNIT 2
UNIT 1
h_3
Z_3
h_2
Z_2
h_1
DATUM
$Z_1 = 0$
BED ROCK
CASE 1
STATIC EQUILIBRIUM
UNIT 3
UNIT 2
UNIT 1
Z_3
Z_2
GRAVEL AQUIFER
CASE 2
WATER MOVING VERTICALLY DOWNWARD
UNIT 3
UNIT 2
UNIT 1
h_3
Z_3
h_2
Z_2
h_1
CONFINED AQUIFER
CASE 3
WATER MOVING VERTICALLY UPWARD

Figure 26. Hydraulic head distribution in three piezometers during three hypothetical flow cases (after Reeve, 1965).

A similar analysis also shows that flow occurs between units #3 and #2 under a unit hydraulic gradient.

The third case represents a water table region underlain by a confined aquifer. At the same time, evaporation is occurring at the land surface. The depth to water is greater in the shallower units. This is, the pressure head increases with depth in the profile. An analysis using hydraulic principles would show that the total hydraulic head decreases with distance above the datum so that water is moving vertically upward in the system below the water table. The last two cases demonstrate an important principle. Namely, when vertical flow occurs, it may be difficult to locate the actual depth to a water table in a moving system using only the data from piezometers.

The cases shown on Figure 26 do not include the effects of lateral flow superimposed on vertical flow. It is known that many field soils exhibit anisotropy. That is, the hydraulic conductivity in the horizontal direction may be many times greater than in the vertical direction. For example, Bouwer and Jackson (1974) observed the ratio of horizontal to vertical hydraulic conductivity to be 7:1 for a finely stratified loamy sand. For such soils, it may be advisable to install more than one array or battery of piezometers in a lateral direction away from a flooding operation. The basal openings of corresponding units in sequential arrays should terminate at the same elevation. Such an arrangement would permit observing lateral head differences resulting from horizontal flow.

A basic problem with piezometers is that a lag may occur in the response of the units to a rapidly changing system and when flow occurs in stratified material. Bouwer and Jackson (1974) presented a time lag equation for piezometers:

$$T_L = \pi r^2 \, Y_0/AK \, Y_0 = \pi r^2/AK \qquad (29)$$

where T_L = time lag constant

r = radius of tube

Y_0 = distance of sudden rise of water level

A = geometry factor

K = hydraulic conductivity.

The value of A is determined from suitable equations, graphs, or tables. It may be shown that the basic time lag manifests the time for which 63 percent of the sudden change in actual water level position has been registered by the water level in the measuring device (Bouwer and Jackson, 1974).

Since T_L is directly porportional to r, more rapid response may be obtained by reducing the area of the opening of the piezometer. Lissey (1967) presents a piezometer design based on this principle.

Observation wells--An observation well consists of an uncased borehole or perforated pipe extending from ground surface into a perched water table. For shallow water tables, observation wells may be installed by hand augers or simple drilling equipment. For deeper wells (e.g., those extending below the main water table), it will be necessary to use standard drilling equipment. Development of cased wells by surging or pumping may be necessary to ensure free flow of water into and out of perforations.

A principal function of observation wells is to permit observation of water table fluctuations. During water-spreading operations, for example, water level changes in observation wells may manifest the arrival of downward flowing effluent. Water levels are measured by means of chalked tape, electric sounders, or air lines. Continuous records of water table fluctuations are possible with the use of automatic water-stage recorders. Water table readings should be referenced to a datum, e.g., the top of the casing, so that levels from a network of wells can be used to construct isopiestic maps.

Under steady flow conditions, an observation well may perform adequately in providing data on the location of the water table. However, during transient vertical flow, reliable estimates are difficult to obtain for reasons stated by Kirkham (1947):

> If there is downward seepage, the bottom of the perforated pipe acts as a sink and the water table is accordingly depressed about the pipe; if there is upward seepage, the bottom of the perforated pipe acts as a source, the water table then mounding up about the pipe.

Saturated Hydraulic Conductivity--

Alternative techniques have been developed for estimating the saturated hydraulic conductivity of vadose zone materials either in the presence or absence of a water table. Detailed reviews of these methods were presented by Bouwer and Jackson (1974) and Bouwer (1978).

Estimates of K within perched groundwater bodies of the vadose zone may be obtained by pumped hole techniques such as the auger hole method, the piezometer method, tube method, well point method, and multiple-well methods (Bouwer and Jackson, 1974). The auger hole, piezometer, and well point methods entail installing a single cavity (cased or uncased) below the water table, lowering the water level in the cavity by pumping or bailing, and measuring the rate of recovery of the water level. Knowing the geometry of the cavity and the known or assumed depth to an impermeable layer, suitable equations, curves, or nomographs may be used to calculate K. In the multiple-well method, water is pumped from one well or set of wells into other wells until the difference in water levels between the wells becomes stabilized. Knowing the geometry of the cavities and flow system, appropriate equations or curves are used to determine K (Bouwer and Jackson, 1974, pp 612-627). All of the above methods, except the tube method, determine K primarily in a horizontal direction.

If the perched groundwater region is extensive containing productive wells, it may be possible to use standard pumping test methods to determine the transmissivity, T, of the formation. Such methods are reviewed in detail by Lohman (1972). Knowing the transmissivity of the system and the thickness of the layer, m, the perching layer, the hydraulic conductivity is estimated by the relationship $T/m = K$. A problem may exist in using this method if leakage beneath the perching layers is substantial.

In many waste disposal operations, it is of value to estimate the saturated hydraulic conductivity in the vadose zone even though a water table is absent. For soils, the saturated K value indicates the potential intake rate. Thus, comparison of K values of soils at a number of sites using the methods discussed below will facilitate locating a disposal operation to minimize deep seepage. Similarly, a range of K values in deeper strata of the vadose zone will indicate the location of possible perching layers. In addition, if hydraulic gradients are assumed to equal unity, the K values represent an upper limit of flux through the vadose zone.

A number of field techniques have been developed for estimating K in shallow soils in the absence of a water table. Determination of the hydraulic conductivity, K, in the absence of a water table generally involves techniques which bring the soil at the measuring point to saturation or near-saturation. Bouwer and Jackson (1974) reviewed five possible methods: (1) the shallow well pump-in method, (2) the cylinder permeameter method, (3) the infiltration gradient technique, (4) the air-entry permeameter technique, and (5) the double-tube method. The shallow well pump-in method measures K mainly in the horizontal direction and is suitable for stoney soils. The cylinder permeameter and infiltration gradient techniques measure K in the vertical direction and are not suitable for stoney soils. The air-entry permeameter measures K in a vertical direction and with care may be used in stoney soils. An important consideration in using these methods is to minimize entrapping air in the pores of the media. Obviously, occluded air will decrease the value of the measured K below the true saturated value. For additional details on these methods, including associated equations and techniques for determining K, the reader is referred to the review of Bouwer and Jackson (1974).

The above methods are suitable primarily for determining K in shallow regions of the vadose zone.

Laboratory and field methods are currently available for measuring the hydraulic conductivity in deeper unsaturated regions of the vadose zone. Laboratory related techniques which have been used to estimate K values from drill cuttings include permeameter tests and empirical relations between the grain-size distribution and K. Permeameter tests are not particularly meaningful because of the disturbance caused by the drilling process. In contrast, certain empirical relationships might exist between grain-size analyses and the hydraulic conductivity. Davis and deWiest (1966, p 375) present a table relating hydraulic conductivity to the dominant size of selected sediments.

The U.S. Bureau of Reclamation (1977) has developed two alternative field techniques for estimating the saturated hydraulic conductivity, K, of unconsolidated unsaturated sediments of the vadose zone. One approach entails pumping water into a borehole at a steady rate such that a uniform water level is maintained within a basal test section. Knowing the dimensions of the hole and inlet pipes, the depth of water, and the constant inflow rate, appropriate equations and curves are consulted to calculate K. For depths less than 40 feet, the hole is cased to the desired depth. A water inlet pipe and a separate water level measurement pipe are then placed inside the casing. Gravel is then poured inside the casing to ensure a gravel pack throughout the test section. Subsequently, the casing is pulled back and a test is initiated. For depths greater than 40 feet, it is necessary to use preperforated casing to isolate the test section. The total open area of the perforations must be known. An observation pipe is placed in the casing on a 6-inch bed of gravel. The use of preperforated casing permits driving the casing in depth-wise increments such that a profile of K values is obtained.

The second method for estimating K is used in the vicinity of a widespread lens of slowly permeable material. The method entails installing an intake well and a series of observation wells. Water is pumped into the well at a steady rate and the water level response in the observation wells is recorded. Appropriate equations and curves are consulted to calculate K.

Weeks (1978) presented a method for measuring vertical air permeability values of layered materials in the vadose zone. Basically, the method entails measuring air-pressure changes in specially constructed piezometers during barometric pressure changes at the land surface. By coupling pressure-response data with auxiliary information on air-filled porosity and numerical solutions of the one-dimensional flow equation, an estimate of the air permeability is obtained. If the material is well-drained and permeable enough that the Klinkenberg effect (see Glossary) is minimal, air permeability values may be converted to the corresponding hydraulic conductivity values. The interested reader should consult the report of Weeks (1978) for additional details.

Flow Rates in Perched Groundwater--

The velocity of water movement in extensive perched groundwater and the total discharge are frequently of interest. These characteristics are estimated from the following forms of Darcy's equation:

$$v = -K \left(\frac{H_2 - H_1}{L}\right) \tag{30}$$

and

$$Q = -KA \left(\frac{H_2 - H_1}{L}\right) \tag{31}$$

where v = specific discharge

K = hydraulic conductivity

H_2 = total head in downstream well

H_1 = total head in upstream well

L = distance between wells

A = cross-sectioned area of aquifer normal to flow direction

Q = total discharge.

The hydraulic conductivity, K, of the perched system could be estimated using the methods discussed above. For an example of the use of these equations, the reader is referred to Bouwer (1978, pp 46 and 47).

Combination of Methods for Observing Saturated and Unsaturated Flow--

A combination of methods will be necessary for monitoring the flux of wastewater from ground surface into a perched groundwater table. Such a system is shown on Figure 27. The tensiometers and electrical resistance blocks could be used to monitor flux, hydraulic gradients, and the direction of water movement in the unsaturated region. The observation well is useful in estimating the response of the water table to recharge. The piezometer nest is used to detect the vertical direction of flow beneath the water table.

Again, it must be kept in mind that observation wells and piezometers will not accurately detect the location of the water table in a fluctuating system. However, by installing an access well and monitoring water content, positional changes in the water table may be monitored via a neutron logger. Although more effective than observation wells, the piezometers in monitoring water-level response, the neutron logger measures water content changes and not energy changes.

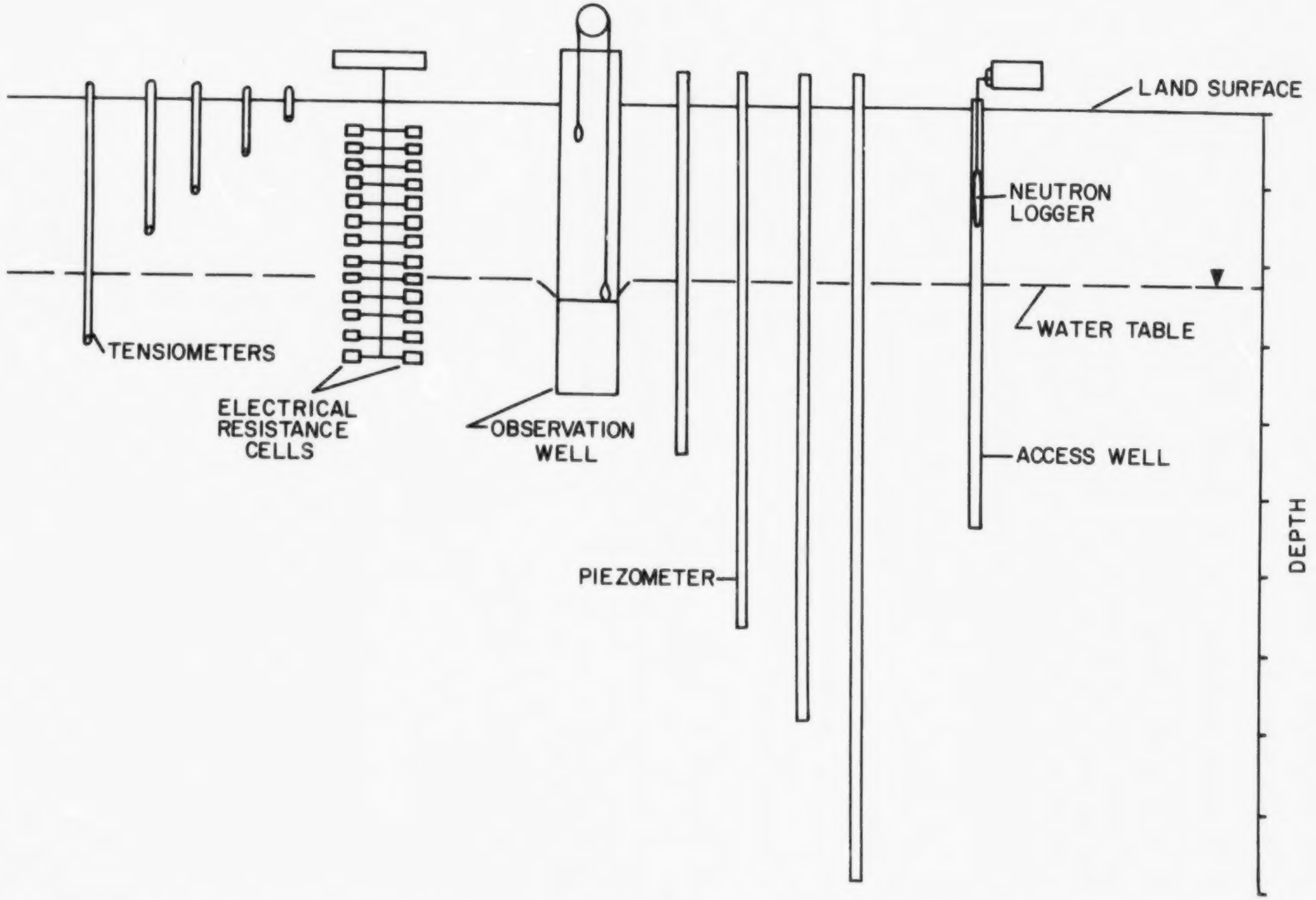

LAND SURFACE
NEUTRON LOGGER
WATER TABLE
ACCESS WELL
DEPTH
TENSIOMETERS
ELECTRICAL RESISTANCE CELLS
OBSERVATION WELL
PIEZOMETER

Figure 27. Array of devices for monitoring water movement in the vadose zone and perched groundwater (after Freeze and Cherry, 1979).

SECTION 6

MONITORING QUALITY CHANGES IN THE VADOSE ZONE

So far in our discussion on monitoring in the vadose zone, we have concentrated on the storage and movement of water, the vehicle for transmitting pollutants. In a quality monitoring program, the major emphasis will be to monitor the spatial and temporal variations of specific water-borne pollutants. In the ensuing discussion, such pollutants are categorized as follows: major inorganic chemical constituents, trace chemical constituents, organic chemical constituents, microbial constituents, and radionuclides. Physical constituents, such as temperature and suspended sediment, will not be considered.

Particular constituents within each of these categories which should be monitored in the vadose zone will obviously be site-specific, i.e., depend upon the waste disposal operation. For example, when monitoring agricultural return flows, specific pollutants may include the nitrogen series, particularly NO_3-N, phosphate, total dissolved salts (TDS), and pesticides. For land treatment sites, monitoring may emphasize bacteria, virus, heavy metals, NO_3-N, and phosphate, TDS, total organic carbon (TOC), biochemical oxygen demand (BOD), and chemical oxygen demand (COD). For sanitary landfills, a range of pollutants in all four categories should be monitored in leachate, with particular emphasis on heavy metals. Finally, for hazardous waste disposal operations, great emphasis should be placed on monitoring the fate of specific organic toxins during flow in the vadose zone.

In the following discussion, reactions affecting the movement of pollutants in the vadose zone are briefly discussed and current methods for vadose zone monitoring are reviewed.

TECHNICAL REVIEW

As with water movement, the present state of knowledge on pollutant movement in the vadose zone is derived from the efforts of specialists working in soils and deeper geological regions. For example, soil chemists and soil microbiologists have examined chemical and microbiological interactions, respectively, in soils. Similarly, geochemists have studied low-temperature chemical reactions in groundwater. Among the references which the interested reader may wish to consult for greater detail on soils/geochemistry relative to pollutant movement are the following: (1) Bohn, McNeal, and O'Connor (1979), a review of the modern concepts of soils chemistry; (2) Alexander (1961), a text on soil microbiology; (3) Hem (1970), a detailed review of water chemistry, including a discussion of chemical constituents in groundwater;

and (4) texts by Davis and deWiest (1966), Bouwer (1978), and Freeze and Cherry (1979) containing chapters on water quality.

Several review papers have also been published in recent years dealing with interactions between water, pollutants, and vadose zone materials. Such papers include those of Ellis (1973), Rhoades and Bernstein (1971), Murrmann and Koutz (1972), McNeal (1974), Pratt et al. (1978), and Chang and Page (1979). In his review paper, Runnells (1976) itemized 11 physical-chemical processes that may operate in the subsurface to purify liquid wastes. The specific processes are: dilution, buffering of pH, precipitation by reaction of wastes with indigenous water or solids, precipitation due to hydrolysis, removal due to oxidation and reduction, mechanical filtration volatilization and loss as a gas, biological assimilation or degradation, radioactive decay, membrane filtration and sorption. Runnells (1976) briefly discussed each of these factors, and the interested reader should consult his paper for details. Fuller (1977) also presented a review article dealing with the movement of selected heavy metals in soils. Among the factors which Fuller (1977) examined relative to the movement of these metals were: soil pH, oxidation-reduction (eH), surface area of soils, pore-size distribution, organic matter, concentration of ions or salts, and presence of hydrous oxides. Pratt et al. (1978) discussed the removal of biological and chemical contaminants by soil systems. The removal process included filtration, adsorption, decomposition, ion exchange, oxidation-reduction, chemical complex formation, chemical precipitation, and other chemical reactions.

In recent years, a number of soil column experiments have been conducted in the laboratory to identify the factors influencing the mobility of pollutants. The column studies of Fuller (1978) and his associates at the University of Arizona on the mobility of trace contaminants from landfill leachate are an example of such laboratory studies. Based on statistical analyses of these laboratory studies, Korte et al. (1976) found that the following factors were dominant in affecting the movement of trace contaminants in soils: soil texture and surface area, percentage of free oxides, and pH.

Field studies have also been conducted to determine the factors affecting the movement of pollutants in vadose zones. Apgar and Langmuir (1971) obtained water samples to a depth of about 55 feet from the vadose zone beneath a landfill in Pennsylvania. (The technique used to obtain these samples was suction cup lysimeters, described later in this report.) Based on their field observations, Apgar and Langmuir (1971) concluded that the following factors were important in attenuating pollutants: dilution and dispersion, oxidation, chemical precipitation, cation exchange, and anion exchange. During field studies to determine chemical changes in water during artificial recharge at a site in Texas, Wood and Signor (1975) examined the following mechanisms: cation exchange, anion exchange, mineral solution, adsorption, desorption, and sulfate reduction. They concluded that at their site, ion exchange and desorption were the major mechanisms affecting water quality in the vadose zone.

Ion Exchange, Cation Exchange Capacity, and Sorption/Desorption Effects in Soils

It is generally recognized by soil scientists that among the factors affecting the mobility of the chemical constituents of water in soils, ion exchange and sorption/desorption are of prime importance. A brief review of these factors is presented in this section. For details, the reader should consult the text of Bohn, McNeal, and O'Connor (1979).

Ion exchange represents an important soil water interaction for altering the composition of the soil solution. Ion exchange is a function of the cation exchange capacity (CEC) of a soil, which in turn is related to colloidal clay minerals, soil organic matter, iron, and aluminum sesquioxides and hydrous oxides.

Clays commonly produce an overall net negative change on their exchange complex. Murrmann and Koutz (1972) pointed out that the cation exchange property arises from the need to balance the negative charge on clay micelles to maintain neutrality. To accomplish this, a swarm of positive ions in the soil solution becomes associated with the negative charge on the exchange complex. These ions are mobile and readily exchange with other cations in the soil solution to maintain chemical equilibrium. This process represents cation exchange. Representative CEC values range from 100 milliequivalents per 100 gms (meq/100 gms) for montmorillonite clays to 5 meq/100 gms for kaolonite type clays.

For most soils, the level of exchangeable cations on the exchange complex is greater than the amounts in the soil solution. Thus, a plot of ionic concentration versus distance from the clay surface (i.e., from the micelle surface) shows a marked drop-off in cation concentration away from the surface, approaching asymptotically to the level in the soil solution. In contrast, the concentration of anions in the vicinity of the micelle is lower than the concentration in the soil solution. In fact, because of this repulsion of anions, the anion concentration in the solution extracted from clay may be greater than in the solution originally added (Rose, 1966), a phenomenon known as negative adsorption.

The ease with which a given ion on the exchange complex exchanges with ions in solution is, among other things, a function of the valence of the ion--the smaller the valence of the ion, the more readily is it exchanged. The ease of replacement of common soil cations is (Rose, 1966): Li > Na > K > Mg > Ca > Ba > Al.

The organic matter fraction of a soil may contribute substantially to the CEC. In fact, additions of organic matter through sludge application may improve the CEC of soils with naturally low CEC (Broadbent, 1973). In addition to contributing a negative charge to the soil exchange complex, organic matter together with the sesquioxides may contribute a positive charge. Apparently, this phenomenon is highly pH dependent.

A complex relationship exists between soluble salts in the soil solution, those on the exchange complex, and salts in the solid (precipitated) phase.

The complexity is increased when it is necessary to account for the effects of soil water. Aspects of this problem will be reviewed later under the discussion of methods to estimate CEC and soluble salts.

In the opinion of Murrmann and Koutz (1972), adsorption of metals onto the surface of soils is the most important process for removing chemicals from wastewater. In contrast to ion exchange in which ions retain their mobility, in adsorption reactions, ions are held so tightly that they become essentially immobile. Exact mechanisms for adsorption are not clear although covalent bonding appears to be important (Chang and Page, 1979). Methods to quantify the extent of adsorption of ions onto solids have been developed. In particular, adsorption isotherms, such as the Freundlich isotherm and the Langmuir adsorption isotherm, are commonly used. Ellis (1973) discussed the use of these isotherms to quantify the adsorption of phosphate, sulfate, boron, and heavy metals by soils.

Rhoades and Bernstein (1971) indicated that the tremendous sorptive area of soils, coupled with high cation exchange capacities, influences the fate of pesticides, radionuclides, nutrient elements, and other inorganic solutes during disposal in soils.

In their artificial recharge studies in Texas, Wood and Signor (1975) concluded that silica was the only constituent affected by desorption. The source of the mobilized silica apparently was not dissolution of quartz or amorphous silica, but desorption of monomeric silica from the aquifer matrix. They concluded that, "The amount of mobilized silica was very large, and the process of desorption should be considered in a predictive model of water quality in areas similar to the one desorbed."

A commonly used measure of the partitioning of ionic species between solid and liquid phases as a consequence of processes such as adsorption is the "distribution coefficient," K_d. This coefficient was defined (Wheeler, 1976) as follows:

$$K_d = \frac{\text{concentration of ionic species on solid phase (gm/gm)}}{\text{concentration of ionic species in soil water (gm/ml)}} \qquad (32)$$

According to Freeze and Cherry (1979), the distribution coefficient "... is a valid representation of partitioning between liquid and solids only if the reactions that cause the partitioning are fast and reversible and only if the isotherm is linear." Wheeler (1976) pointed out that K_d varies from zero for nonreactive species such as chloride to 10^5 for some actinides (the actinides include uranium, plutonium, and americium).

Wheeler presented the following expression to describe the rate of flow of an ionic species relative to water during partitioning:

$$\frac{v_i}{v_w} = \frac{1}{1 + \frac{D_b}{\theta} K_d} \qquad (33)$$

where v_i = pore velocity of adsorbed species

v_w = pore velocity of migrating water

θ = volumetric water content

D_b = bulk density of soil.

This equation is applicable to both saturated and unsaturated media. Factors affecting the use of equation 33 are: (1) the assumption that reactions are fast and reversible may not be satisfied, (2) the adsorption isotherm may not be linear, (3) the distribution coefficient may not be constant with varying soil-water contents, and (4) concentrations of other ionic species may affect the magnitude of K_d (Wheeler, 1976).

Laboratory Methods: Soluble Salts, CEC, Exchangeable Ions, Sorptive Capacity, Column Studies, and Batch Tests

Soluble Salts--

As defined by Bower and Wilcox (1965), soluble salts refer to the soil constituents of an inorganic nature that are appreciably soluble in water. The following steps are usually taken in an analysis for soluble salts: preparing a sample by bringing the soil-water content to some prescribed value, extracting the sample, and measuring or analyzing the salt content of the extract.

The soil-water content for extracting soluble salts is a matter of concern. Saturation extracts, for example, are commonly used by agriculturists to relate soluble salts to field moisture range. A problem with diluting the sample when obtaining saturation extracts is that, while the concentration of some ions increases on dilution, the concentration of others may decrease (Reitemeier, 1946). For example, as water is added to the soil, precipitated calcium carbonate or gypsum may gradually dissolve, releasing calcium and possibly magnesium ions into solution. Concurrently, the additional concentrations of calcium and magnesium may displace sodium on the exchange complex, increasing the sodium level in the soil solution. Overall, calcium, magnesium, and sodium will increase in the soil solution. In contrast, chloride and nitrate will decrease in concentration because of dilution and negative adsorption. Such trends have been experimentally observed during laboratory studies (Reitemeier, 1946). Reitemeier recommended that in arid soils, dissolved ions should be determined at or near the water content at which the results are to be applied. Consequently, the soil solutes of samples from deeper horizons of the vadose zone should be extracted at the prevailing soil-water pressure.

Pressure membrane apparatus may be used to extract soil solution samples in the dry range (Richards, 1954). Thus, if a soil-water characteristic is available for the sample, together with gravimetric water content data, the equivalent soil-water pressure can be determined. This pressure is then applied to the soil sample in the membrane. A problem with this technique is that, inasmuch as the solution is extracted mainly from the larger pores, the chemical composition may not be the same as that for solution from the smaller sequences (Rhoades and Bernstein, 1971).

For surface soils, other factors should be examined in the determination of soluble salts. For irrigated soils or on land disposal operations subject to wetting and drying, the water content of soils may range over large values because of evaporation, so that conversion to the field water content value may not be meaningful (Pratt, Jones, and Hunsaker, 1972). In this case, the saturation extract technique is recommended (Rhoades and Bernstein, 1971). Water content by this method represents about twice that at field capacity. Therefore, the salt content extracted from the saturated sample is about one-half the concentration at field capacity. Bower and Wilcox (1965) explain in detail the procedure to obtain a saturated extract. Briefly, deionized water is mixed into a weighed oven-dry sample of soil until the soil glistens and no free water has collected on the soil surface. If this end point is sustained for an hour or more, the sample is placed in a Buchner funnel. Vacuum is applied and the filtrate collected for analyses. Total salt concentration of the extract may be estimated by measuring the specific electrical conductance and using the relationship (Bower and Wilcox, 1965): salt concentration (mg/l) = 640 x electrical conductivity, mmho/cm (Ayers and Westcot, 1976).

If detailed information is required on specific ionic constituents in extracts, chemical analyses using procedures in "Methods of Soil Analyses" (Black, 1965) may be used. According to Bower and Wilcox (1965), the principal ions of importance in soils studies are: Ca^{++}, Mg^{++}, K^+, Na^+, $CO_3^=$, $SO_4^=$, CL^-, and boron, as well as the nitrogen series.

Cation Exchange Capacity and Exchangeable Ions--

The sum of individual exchangeable basis of a soil sample is equal to the cation exchange capacity. Alternatively, the CEC may be obtained directly using methods detailed by Chapman (1965). Briefly, the exchangeable cations in a soil sample are replaced by either ammonium acetate or sodium acetate, and the amounts of ammonium and sodium ion adsorbed are determined. A problem may develop with the use of ammonium ion because this ion becomes strongly adsorbed on some clays. Cation exchange capacity is expressed as milliequivalents per 100 gms (meq/100 gms) of sample.

Sorptive Capacity: Specific Surface--

The cation exchange capacity and specific surface together govern the sorptive characteristics of a soil. Mortland and Kemper (1965) discussed the principles of adsorption relating to the specific surface of clays and reviewed a number of adsorption isotherms. A method for determining specific surface based on sorption of ethylene glycol was presented.

Column Studies and Batch Tests--

An approximate idea of the attenuating properties of vadose zone materials for specific pollutants can be obtained from laboratory column studies. In practice, cylindrical columns are packed with vadose zone samples (core samples could also be used) and subsequently flooded with wastewater from the disposal site. Samples of column effluent are collected and analyzed and the breakthrough of particular constituents is determined. Advantages of laboratory column tests include the following: (1) the method is simple, (2) results are related to a specific mass of material, and (3) results may generally be obtained in a short time. Disadvantages include: (1) if disturbed samples are used, water movement in the column may differ from in-place flow, (2) flow along the column walls may occur, and (3) flow rates may differ from in-place rates.

Batch testing is an alternative method for estimating the attenuation of water-borne pollutants as a result of interactions in the vadose zone. Batch tests consist of placing fragmented samples of vadose zone material in flasks together with a measured aliquot of wastewater. The flasks are shaken via a mechanical shaker for a given period of time. The fluid is subsequently drained from the flasks and analyzed for constituents of interest. Advantages of batch tests include the following: (1) the method is simple and inexpensive, (2) results are related to a specific mass of material, (3) results are obtained in a short time, and (4) results may be used to prepare adsorption isotherms or selectivity coefficients in ion exchange reactions (Freeze and Cherry, 1979).

Disadvantages of batch tests include the following: (1) results may be affected by sample disturbance, (2) flow conditions differ from those in place, and (3) samples are exposed to the air, i.e., adsorptive capacity of oxidized material may differ from reduced material, affecting the transferability of results to field cases (Freeze and Cherry, 1979).

Attenuation of Specific Constituents in the Vadose Zone

Major Chemical Constituents--

Certain wastewaters such as landfill leachate may contain excessive concentrations of the major chemical constituents, including calcium, magnesium, sodium, nitrate, chloride, sulfate, phosphate, and bicarbonates. The fate of both calcium and magnesium is dependent upon precipitation and affected by sorption on the cation exchange complex of clays. Generally, calcium and magnesium will precipitate during reactions with bicarbonate and sulfate. McNeal (1974) discusses precipitation reactions in detail. Precipitation of carbonates occurs as a result of the concentration of the soil solution, although the solubility is affected by the partial pressure of CO_2 in the gas phase and the salt concentrations of the solution. McNeal (1974) illustrated the use of the "Langelier saturation index" to predict the approximate amounts of calcium carbonate precipitating from waters.

Next to precipitation in the carbonate form, the precipitation of calcium in the form of $CaSO_4$ (gypsum) is important as a process for removing calcium from solution. It is possible to predict the precipitation of gypsum using Debye-Huckel Theory.

The solubility of $MgCO_3$ is also influenced by the presence of CO_2, and free movement within the vadose zone will occur until the pH elevates.

Sodium and potassium salts are soluble and mobile unless concentrations are increased to several thousands of parts per million.

In addition to precipitation, calcium and magnesium mobilities will be limited by exchange reactions in clays. For this case, calcium and magnesium will be in competition with sodium for exchange sites. The relative degree of sodium adsorption is expressed by the adjusted sodium adsorption ratio, adj SAR, defined by Ayers and Westcot (1976) as:

$$\text{adj SAR} = Na^+/((Ca^{++} + Mg^{++})/2)^{1/2} \, (1 + (8.4 - pHc)). \qquad (17)$$

Terms of this equation were defined earlier.

The mobility of nitrogen compounds in vadose zone materials is primarily related to the oxidation-reduction potential. Thus, if the system is aerobic, nitrification of organic-N sources occurs readily, producing NO_3-N as an end product. Nitrate-N is highly mobile, moving readily with the soil solution into the lower vadose zone and ultimately into groundwater. Under anaerobic conditions, nitrification is inhibited and the NH_4-N form predominates. Ammonium-N is attenuated in soils with clays or organic materials by two mechanisms: sorption of the positively charged ammonium ion on the clay-organic exchange complex, and fixation of the ammonium ion within the crystal lattice of clay minerals. The latter mechanism is particularly pronounced in clays with a 2:1 silica-alumina ratio (Nommik, 1965). Ammonia sorption on the exchange complex is affected by the presence of other cations, such as calcium and magnesium which compete for available sites.

In soil systems in which wastewater is applied intermittently, nitrification of sorbed NH_4-N occurs during drying cycles (Bouwer, 1978). Consequently, during subsequent flooding cycles, NO_3-N may be leached into the lower vadose zone.

Under anaerobic conditions, nitrogen may be lost through denitrification. In particular, denitrification occurs quite readily as a result of the activity of heterotrophic bacteria, which convert NO_3-N to volatile gasses such as N_2O and N_2 via a number of intermediate compounds. An organic substrata is necessary as an energy source for denitrifying microorganisms. During controlled field studies, Rolston and Broadbent (1977) noted that denitrification was also affected by soil-water content, soil temperature, and by the presence or absence of a growing crop in the soil.

The mobilities of sulfate and bicarbonate are linked to reactions with calcium and magnesium. In addition, sulfate may be sorbed to a minor extent in the aquifer matrix and be retained by Fe hydrous oxides (Keeney and Wildung, 1977). The latter reaction, however, requires low pH. The presence of organic acids in leachate migrating into the vadose zone may result in reducing sulfate mobility. Reduction of sulfate under anaerobic conditions may lead to the formation of H_2S and eventually insoluble sulfides such as ZnS and FeS_2. An increase in bicarbonate often accompanies an increase in sulfate (Wood and Signor, 1975).

Factors influencing the solubility of iron include pH, the redox potential (eH), and the dissolved CO_2 and sulfur species (Hem, 1970). The reduced form, i.e., the ferrous ion, is soluble and mobile. The oxidized form, or ferric ion, forms relatively insoluble precipitates with sulfur and carbonate species. Hem (1970) presented a pH-Eh diagram which demonstrates the conditions under which iron solubility is very low.

Phosphate retention and mobility were discussed by Keeney and Wildung (1977). Under acid conditions, phosphorus is sorbed on the surface of Fe and Al containing minerals. Organic acids may have a local effect on lowering pH values, promoting the above effect. For alkaline conditions, the sorption of P on $CaCO_3$ or formation of Ca phosphate minerals may occur (Keeney and Wildung, 1977). Phosphate retention on clays and hydrous oxides may also be important.

Trace Contaminants--

Among the general mechanisms associated with the removal of trace metals, it appears that adsorption is of prime importance. In reviewing the results of recent laboratory studies on metal attenuation, Chang and Page (1979) indicated:

> ... strong adsorption of trace metal ions occurred at the surface of amorphous iron and manganese oxides and aluminum minerals.... Other soil properties such as texture and cation exchange capacity did not appear to significantly influence soil adsorption characteristics. Unlike the electrostatic cation exchange reactions, covalent bonding-induced adsorption generally is more specific and the reactions are not easily revised by the presence of other cations.... Even trace elements which usually form anions in aqueous solution may be effectively adsorbed or rapidly converted into insoluble forms.

Murrmann and Koutz (1972) also compared the relative effects of cation exchange and adsorption for removal of heavy metals in wastewaters. They indicated that in contrast to cation exchange, "... soil has a capacity to retain heavy metals so tightly that they can be replaced only with difficulty."

Based on results of laboratory studies, Korte et al. (1976) developed figures on the relative mobility of 11 trace contaminants in the 10 most prominent soils orders of the United States. These figures are reproduced on Figures 28 and 29.

Recently, Fuller (1977) published a comprehensive report on the movement through soils of the following elements: arsenic, beryllium, cadmium, chromium, copper, cyanide, lead, mercury, selenium, and zinc. The interested reader should review this report for details. According to Fuller, the principal mechanism for attenuating arsenic is adsorption by soil colloids. However, if waterlogging should occur during migration of leachate, reducing conditions will favor the mobilization of arsenic. Finally, according to Fuller, "At the low concentrations usually found in wastewaters, landfill leachates, and other aqueous waste streams, As probably will not precipitate in

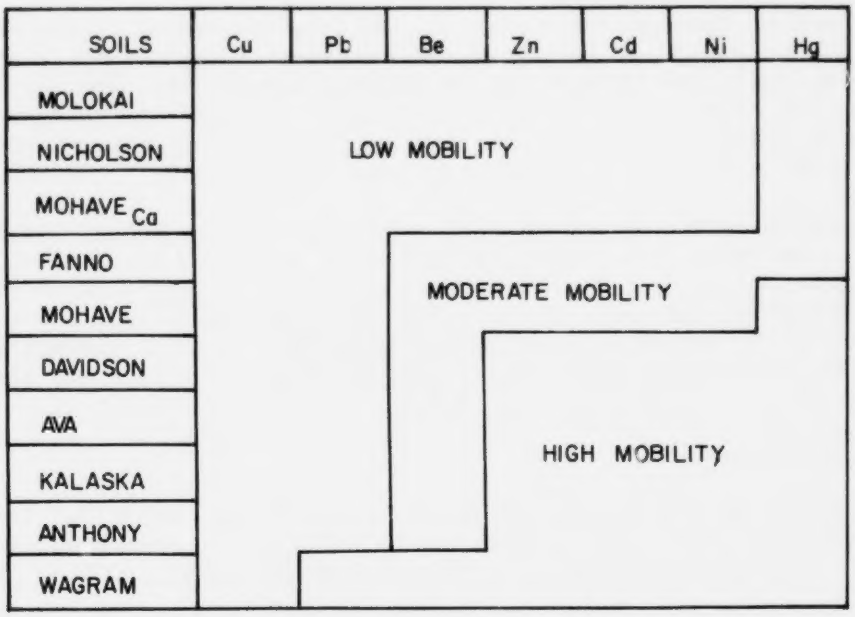

SOILS
Cu
Pb
Be
Zn
Cd
Ni
Hg
MOLOKAI
NICHOLSON
MOHAVE Ca
FANNO
MOHAVE
DAVIDSON
AVA
KALASKA
ANTHONY
WAGRAM
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

Figure 28. Mobility of Cu, Pb, Be, Zn, Cd, Ni, and Hg in 10 soils (after Korte et al., 1976).

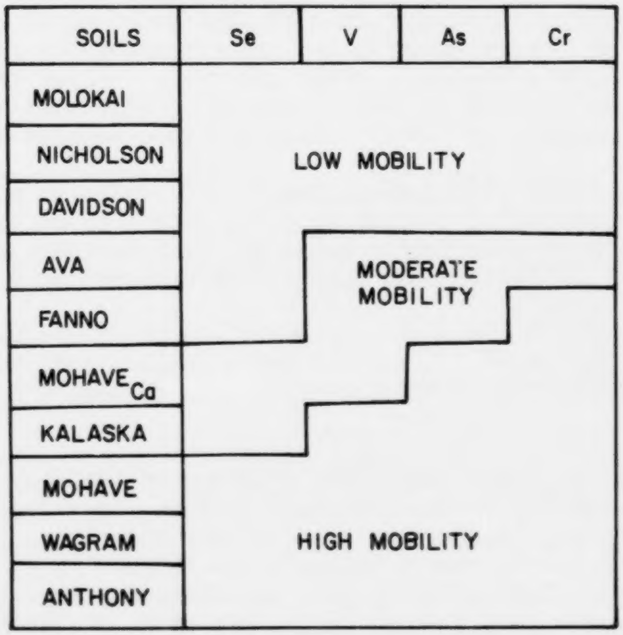

SOILS
Se
V
As
Cr
MOLOKAI
NICHOLSON
DAVIDSON
AVA
FANNO
MOHAVE Ca
KALASKA
MOHAVE
WAGRAM
ANTHONY
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

Figure 29. Mobility of Se, V, As, and Cr in 10 soils (after Korte et al., 1976).

soils except possibly as an impurity in phosphorus compounds formed over a long period of time."

The complex chemistry of selenium is described by Fuller. The behavior of selenium is closely related to that of sulfur in acid formation and other properties. Based on experimental studies at the University of Arizona, Fuller concluded that other factors being equal, selenium is less mobile in acidic than in neutral or alkaline soils.

Regarding the mobility of zinc, Fuller (1977) indicated that Zn^{2+} forms slowly soluble precipitates with carbonate, sulfides, silicate, and phosphate ions. Sulfide values may increase in leachate if waterlogging occurs, promoting precipitation of zinc (and other cationic heavy metals). Zinc is also strongly sorbed on the exchange complex of soil.

Unlike other members of the halogen group, fluoride compounds tend to be rather insoluble (Hem, 1970). The mobility of fluoride in leachate may be limited by the formation of fluorite (CaF_2) with a solubility product of $10^{-10.57}$ (Hem, 1970). High concentrations of calcium in leachate would tend to favor fluorite formation in spite of common ion effects.

Reactions of strontium in water are similar to those of calcium. Strontianite, formed by the reaction of strontium and bicarbonate, is slightly less soluble than calcite (Hem, 1970). In addition, relatively insoluble strontium sulfate may be formed in sulfate rich waters (Davis and deWiest, 1966). Both reactions may occur, limiting the mobility of strontium.

Copper appears to be strongly complexed to organic matter (Fuller, 1977); consequently, copper-organic chelates may form in leachate which may be soluble and mobile. The formation of hydrous oxides of Mn and Fe provides the main control in the immobilization of copper. Hem (1970) reported that copper solubility is generally lower in reducing systems than in oxidizing systems, particularly if reduced sulfur species are present. Reducing or anaerobic conditions could exist if saturation develops. Reduction of sulfate, present in leachate, would then lead to precipitation of copper [as well as iron, zinc, cadmium, lead, and mercury (Fuller, 1977)].

Both nickel and cobalt are strongly adsorbed by iron and manganese oxides (Hem, 1970). The low solubility of $CaCO_3$ may be an important factor in limiting cobalt concentrations in solutions in which HCO_3^- concentrations are high. According to Hem (1970), there are no effective solubility controls over molybdenum concentrations in water. Consequently, the mobility of the anionic form, molybdate, will probably be high.

Jurinak and Santillan-Medrano (1974) examined the transport of lead and cadmium in soils. Processes considered to be important in governing cadmium and lead movement in soils included the following: precipitation and dissolution, ion-pair formation, pH flux, cation exchange, and adsorption. Jurinak and Santillan-Medrano (1974) concluded that the principal mechanism regulating Pb solubility in noncalcareous soils was precipitation of the forms: $Pb(OH)_2$ and $Pb_5(PO_4)_3OH$. In calcareous soils, $PbCO_3$ also precipitates. "Calcareous soils appear to be an excellent sink for Pb^{2+} ions."

Jurinak and Santillan-Medrano found that the solubility of cadmium in soils is about 100 times greater than lead in the pH range 5 to 9. "The data suggest that particularly at low concentrations, the adsorption of Cd by the soils is a more important mechanism in retention than in the case of Pb where precipitation of slightly soluble compounds regulates solubility."

Pratt et al. (1978) indicated that in solution, mercury exists in inorganic form as a monovalent or divalent ion and as complex ions or ion pairs. Regarding mobility of mercury, these authors stated, "The concentrations of Hg in soil solutions are governed by ionic adsorption by organic and inorganic materials and by the low solubilities of Hg as phosphate, carbonate, and sulfide."

Alesii (1976) studied the mobility of simple and complex forms of cyanide in soil columns. He observed, "Cyanide as $Fe(CN)_6^{3-}$ and CN^- in water were both found to be very mobile in soils. Soil properties, such as low pH, percentage free iron oxide and Kaolin, chloride, and gibbsite type clay (high positive charges) tended to increase the attenuation of cyanide. High pH, presence of free $CaCO_3$ (high negative charge), low clay content, and montmorillonite clay tended to increase the mobility of the cyanide forms." When an abundant energy source is available, microorganisms volatilize cyanide into less harmful forms of nitrogen. However, as noted by Fuller (1977), anaerobic conditions inhibit the microbial degradation of a cyanide. In a survey of municipal wastewaters in California, Pratt et al. (1978) found that the maximum cyanide level was 0.16 mg/l. They concluded that during recharge, cyanide levels would be reduced well below the recommended limit of 0.2 mg/l.

Organic Constituents--

A problem in specifying the mobility of specific organic pollutants is that quantitative studies have only recently been reported. One problem is that analytical procedures to identify organic constituents are still being developed. "Methods of sampling and analyzing for most organics are in their relative infancy compared to those used for other pollutants" (U.S. EPA, 1979a).

Two alternative methods which may have applicability in characterizing organic pollutants during flow in the vadose zone have been proposed by the U.S. EPA (1979a). One method is quantitative, employing GC/MS (gas chromatography/mass spectrometry). The other method provides for "screening" to detect the presence or absence of organic toxins by GC/MS, followed by extra GC or LC (liquid chromatography) quantification of identified pollutants.

Recently, Leenheer and Huffman (1976) described the development of the dissolved organic carbon (DOC) technique for fractionating organics into hydrophobic and hydrophilic components using macroreticular resins. The technique was applied to several natural waters. This technique has advantages over other methods for concentrating organics, such as activated carbon. For example, Robertson, Toussaint, and Jorque (1974) reported that only 10 percent of organics present in groundwater beneath a landfill in Oklahoma were identified using carbon adsorption followed by carbon chloroform and carbon alcohol extraction.

The principal mechanisms for attenuating organic pollutants are adsorption and decomposition (Pratt et al., 1978). As an example of the importance of adsorption, Robertson, Toussaint, and Jorque (1974) observed that polychlorinated biphenyls (PCBs) tend to be strongly adsorbed in soils. Leenheer and Huffman (1976) indicated that both hydrophobic and hydrophilic organics may be sorbed by sediment. Chang and Page (1979) discussed adsorption of organic substances on soils in some detail:

> The adsorption of dissolved organic substances usually is complicated by the chemical characteristics (i.e., molecular weight, structure and the presence of various functional groups) of each organic compound.... Since many organic substances tend to form negatively charged colloids in aqueous solution, the most likely sites for adsorption to take place would be the positively charged edges of clay minerals. These organic substances also form organometallic complexes which may be adsorbed as neutral or positively charged molecules.

Biodegradable organics are decomposed by microorganisms rendering potentially harmful constituents into gases. Davis (1956) reported that microbiologists have observed the utilization of hydrocarbons by certain bacteria, actinomycetes, filamentous fungi, and yeasts. In general, however, hydrocarbons are not as readily decomposed as carbohydrates, proteins, or fats. Furthermore, the cyclic hydrocarbons are less susceptible to microbial decomposition than are the aliphatic hydrocarbons. Chlorination of wastewaters prior to recharge may destroy both pathogenic organisms and microorganisms responsible for decomposition (Pratt et al., 1978).

The pH may also be a factor in the mobility of organics. For example, Leenheer and Huffman (1976) noted the formation of an organic precipitate upon acidification of a groundwater sample from an oil shale area near Rock Springs, Wyoming. Raising the pH dissolved the precipitate. The pH of leachate during aerobic decomposition within a sanitary landfill may be low enough to cause the flocculation of certain organics. In the underlying media, flocs would then be filtered out. However, in alkaline soils, the pH would eventually elevate to a point in the flow system at which organic flocs would dissolve.

In studies on leaching of spent oil shale, Schmidt-Collerus (1974) noted that the solubilization of polycondensed organic matter was enhanced by the presence of seepage water high in TDS. Possibly, the flux of organics in wastewater would be accelerated in a similar fashion if the initial TDS were high.

Microorganisms--

According to Gilbert et al. (1976), bacteria are removed at the soil surface by filtration, sedimentation, and adsorption. Virus removal occurs mainly by adsorption, which increases with decreasing pH. Other factors listed by Gilbert et al. as important in attenuating bacteria and virus are: salt concentration, pH, organic matter, soil composition, infiltration rates, and climatic conditions. Survival and movement of microorganisms within a soil relate to soil moisture content, temperature, pH, nutrient availability, and antagonisms. The presence of excessive salt levels in wastewater, coupled with the presence of high levels of toxic substances (e.g., trace contaminants, pesticides), may limit the migration of bacteria and virus to a small segment of the vadose zone beneath a landfill.

Specific factors affecting the movement of virus in soils are summarized in Table 4. Chang and Page (1979) cautioned that immobilization of viruses by soil should not be equated with virus inactivation. "Many adsorbed virus particles have been demonstrated to be infectious for significant periods of time. Viruses immobilized by soil adsorption may also become desorbed when the chemical composition of the percolating wastewater is changed."

Pesticides--

Factors influencing the fate and behavior of pesticides in soil systems include: (1) chemical decomposition, (2) photochemical decomposition, (3) microbial decomposition, (4) volatilization, (5) plant or organism uptake, and (6) adsorption-desorption (Bailey and White, 1970). The last item, adsorption-desorption, directly or indirectly influences the magnitude of the other five factors and is considered to be the prime factor governing the interactions between pesticides and soil colloids. Leonard, Bailey, and Swank (1976) classified organic pesticides as ionic or nonionic. In turn, ionic pesticides are subclassified as cationic (paraquat, disquat), basic (s-triazones), and acidic (benzoic acids, phenols, picolinic acid). Nonionic pesticides include chlorinated hydrocarbons and organophosphates. Cationic pesticides are retained tenaciously on the exchange complex. Changes in soil pH have a profound but complex effect on pesticide forms. For example, a decrease in pH increases the molecular form of an acidic pesticide but increases the conjugate acid form of the base (Leonard, Bailey, and Swank, 1976). These changes will modify adsorption-desorption characteristics and the mobility of pesticides. Leonard, Bailey, and Swank also reviewed the effect of clay on sorption-desorption properties and pointed out that organic matter greatly increases the sorptive tendency of soils.

Temperature effects adsorption-desorption of pesticides. For example, an increase in temperature decreases adsorption and promotes desorption (Leonard, Bailey, and Swank, 1976). Temperature also affects sorption through effects on solubility and vapor pressure.

Regarding soil moisture effects, Leonard, Bailey, and Swank indicated, "A decrease in soil moisture (a) causes an increase in concentration per unit volume, ... and may increase surface acidity, and, thus increases adsorption; and (b) causes a decrease in competition with water for adsorption sites, which should increase adsorption." Furthermore, when the pesticide concentrates to the solubility product, crystallization will result.

Pesticide degradation will occur in soils. Photodecomposition will reduce pesticide levels near the soil surface but may be inconsequential with depth. Chemical degradation is a complex phenomena, related to pH, redox potential, and surface acidity.

TABLE 4. FACTORS THAT INFLUENCE THE MOVEMENT OF VIRUSES IN SOIL (after U.S. Environmental Protection Agency, U.S. Army Corps of Engineers, and U.S. Department of Agriculture, 1977).

Factor	Remarks
Rainfall	Viruses retained near the soil surface may be eluted after a heavy rainfall because of the establishment of ionic gradients within the soil column.
pH	Low pH favors virus adsorption; high pH results in elution of adsorbed virus.
Soil composition	Viruses are readily adsorbed to clays under appropriate conditions and the higher the clay content of the soil, the greater the expected removal of virus. Sandy loam soils and other soils containing organic matter also are favorable for virus removal. Soils with a low surface area do not achieve good virus removal.
Flow rate	As the flow rate increases, virus removal declines, but flow rates as high as 32 ft/d (9.8 m/d) can result in 99.9-percent virus removal after travel through 8.2 ft (2.5 m) of sandy loam soil.
Soluble organics	Soluble organic matter competes with viruses for adsorption sites on the soil particles, resulting in decreased virus adsorption or even elution of an already adsorbed virus. Definitive information is still lacking for soil systems.
Cations	The presence of cations usually enhances the retention of viruses by soil.

Microbial decomposition of pesticides depends on such factors as "microbial population ecology, soil moisture and temperature, organic matter content, pH, redox potential, pesticide concentration, availability for degradation, and nutrient concentration and availability" (Leonard, Bailey, and Swank, 1976). In addition to decomposition by microbial activity, pesticide compounds may be taken up by plants.

Volatilization of pesticides will reduce their concentration in the soil. Volatilization depends on such factors as temperature and soil moisture, as well as on the vapor pressure of specific compounds.

Bailey and White (1970) listed the following factors as being most significant in governing the leaching and movement of pesticides in soils: adsorption, physical properties of the soil, and climatic conditions. The effects of adsorption are reviewed above. Regarding soil physical properties, Bailey and White indicated that pesticides are leached to a greater degree in light-textured soils than in heavier-textured soils. The porosity of soils may have an effect on diffusion rates of volatile pesticides. Air diffusion plays a prominent role in the eventual loss of pesticides from soil due to volatilization. Pore-size distribution of soils affects the rate at which the water infiltrates and moves through a soil.

For a given soil type, leaching of pesticides is increased with an increase in the amount and frequency of rainfall. The same relationship holds for irrigation. Evapotranspiration will tend to increase the concentrations of pesticides at the soil surface.

Radioactive Wastes

The principal mechanisms relied upon for the attenuation of radionuclides in the vadose zone are ion exchange and adsorption. Consequently, the presence of clays, organic matter, and hydrous oxides at disposal sites is of paramount importance in reducing the potential for groundwater pollution for radioactive wastes. The "distribution coefficient," K_d, is a commonly used measure of the ability of a solid matrix to retard the movement of radionuclides (and other solutes) via sorption effects.

Borg et al. (1976) reviewed the chemical and physical factors affecting the measured values of the distribution coefficient for radionuclides. They stated:

> Among other variables, mineralogy, particle size, nature of solution, and chemical nature of radioactive species are important.... In general, a decrease in particle size results in an increase in K_d. Fresh silicate rocks have lower K_d's than their altered counterparts. For the same reasons, old fractures absorb more than fresh fractures in a given rock. The sorption of Cs and Sr is greater than that of Ru and Sb and is probably related to the fact that the latter two elements form anionic complexes which do not readily take part in ion exchange processes. Cs in clay-rich rocks often is sorbed more than Sr because of lattice shrinkage that traps the larger ion. Pu forms a positively charged polymer that is highly sorbed in the pH range of 2 to 8.

Regarding the presence of ions in the soil solution competing with radionuclides for sorptive sites, Borg et al. (1976) cited the work of Nelson (1959). In particular, Nelson (1959) conducted batch and column experiments on the sorption of Sr on Hanford soil in the presence of cesium, sodium, barium, calcium, magnesium, and aluminum. He found that water of hydration was a significant factor in that cations with the combination of highest valence and lowest hydrated radii provided the greatest competition with strontium for sites. The work of Kokotov, Popova, and Urbanyuk (1961) was also cited by Borg et al. (1976). Kokotov, Popova, and Urbanyuk (1961) measured K_d values for Sr and Cs in Russian soils. For Sr, the order of effectiveness in lowering the K_d (i.e., increasing the mobility) was $Sr^{2+} > Ca^{2+} > Mg^{2+} > K^+ > NH_4^+ > Na^+$. For Cs, the order of effectiveness in lowering K_d values was $Cs^+ > Rb^+ > NH_4 > K^+ > H^+$.

The effect of soil pH on distribution coefficients of various radionuclides was examined by several investigators. The basis for their studies was that changes in pH affect cation exchange properties of soils and concomitantly K_d values. In particular, increasing the soil pH changes cation selectivity by increasing both the cation exchange capacity and the preference of the exchange complex for polyvalent over monovalent ions (Bohn, McNeal, and O'Connor, 1979). Rhodes (1957b) evaluated the effect of pH on the uptake of the following polyvalent radionuclides: plutonium, cerium, zirconium, yttrium, and ruthenium, as well as monovalent cesium and divalent strontium. The soil used in his tests was from Hanford, Washington. Uptake of the polyvalent species exhibited a maximum between about pH 4 and pH 8. Above pH 8, a region of reduced uptake was observed, persisting up to pH 11. Rhodes (1957b) noted that the effect of increasing pH on uptake of polyvalent radioisotopes is complicated by two possible consequences of changing pH: (1) changing ion species, and (2) formation of polymers or colloids with changing pH.

For example, in an earlier paper, Rhodes (1957a) concluded that plutonium forms a positively charged polymer between pH 2 and pH 8 which is highly sorbed. Apparently, because of a change in the nature of the polymer, sorption of plutonium decreases above pH 8, reaching a minimum of about pH 12.

In his experiments using monovalent cesium, Rhodes (1957b) observed that pH had a negligible effect on the uptake of this ion when HCl was used to adjust the pH of the solution. However, when NaOH was used to adjust the solution pH, a marked reduction in the distribution coefficient was noted. Apparently, sodium ions competed with cesium ions for exchange sites. Experimental results with strontium indicated that when the pH was adjusted with either NaOH or HCl, the distribution coefficient of strontium increased from about pH 4 to about pH 10. Above pH 10, uptake was reduced by the presence of large sodium ion concentrations.

Alternative combined forms of radionuclides may have an effect on distribution coefficients and associated relative velocities. For example, Jakubick (1976) determined that the velocity of PuO_2 in soils is about 100 times faster than the velocity of $Pu(NO_3)_4$.

FIELD METHODS FOR MONITORING POLLUTANT MOVEMENT IN THE VADOSE ZONE

Source monitoring, a primary activity of a monitoring program, is not covered in this section. Instead, the interested reader is referred to the comprehensive methodology of Huibregtse and Moser (1976). For convenience, field monitoring in the vadose zone will be categorized as follows: (1) indirect methods, (2) direct methods for solids sampling, (3) direct methods for solution sampling in unsaturated media, and (4) direct methods for sampling saturated regions.

Indirect Methods

Two alternative properties of a wastewater which could be used to estimate pollutant mobility in the vadose zone are temperature and electrical conductivity. Temperature could serve as an indicator of wastewater movement provided the source were at either an elevated or lowered temperature, and if the disposal site were underlain by shallow perched groundwater. Insertion of sensitive thermistors into the soil and groundwater system could possibly monitor the spread of the source. The limitation of this method is that the temperature wave would be dampened by contact with soil and groundwater. That is, the extent of the plume could be poorly defined.

Resistivity Methods for Soil Salinity--

An indirect property which has been extensively used to characterize soil salinity and to delineate the areal distribution of shallow pollution plumes is resistivity, or its inverse property, electrical conductivity. As pointed out by Rhoades and Halvorson (1977), most soil minerals are insulators. Consequently, electrical conduction in saline soils is mainly through the pore water containing dissolved electrolytes. Exchangeable cations do not contribute extensively to electrical conduction in nonsodic soils because they are not present in abundance and they are less mobile than the soluble electrolytes. According to Rhoades (1979a), the electrical conductivity of a saline soil, EC_a, depends primarily on the electrical conductivity of the liquid, EC_w; on the volumetric water content, θ; on the tortuousity, T; and on the extent of surface conductance, EC_s. For a given soil, the specific conductance of a saturation extract, EC_e, is uniquely related to EC_w. For simplicity, a relationship between EC_w and EC_a is obtained at a uniform water content value, θ, in order to standardize θ and T effects. In practice, the standard water content is taken to be that at field capacity. The relationship between EC_a and EC_e then becomes:

$$EC_a = A\,EC_e + B \qquad (34)$$

where $B = EC_s$

A = slope of the EC_a versus EC_e line.

The four-electrode method--Rhoades and Halvorson (1977) presented three methods for determining EC_a and three methods for establishing EC_e versus EC_a calibrations. One method for measuring soil electrical conductivity in-situ uses the Wenner four-probe array (see Figure 30). This method is a common surface resistivity technique used by geophysicists (Zohdy, Eaton, and Mabey, 1974). Basically, the method entails placing four electrodes into the soil surface at a constant distance "a" apart. An electrical current is applied to the outer two electrodes and the resultant electrical resistance is measured as a potential drop across the inner two electrodes. Resistance is measured via a resistance meter. The apparent bulk soil conductivity, EC_a, is calculated by the following equation:

$$EC_a = \left(\frac{5.222}{a}\right)\frac{f_t}{R_t} \qquad (35)$$

where R_t = measured resistance in ohms at temperature t

a = spacing of electrodes

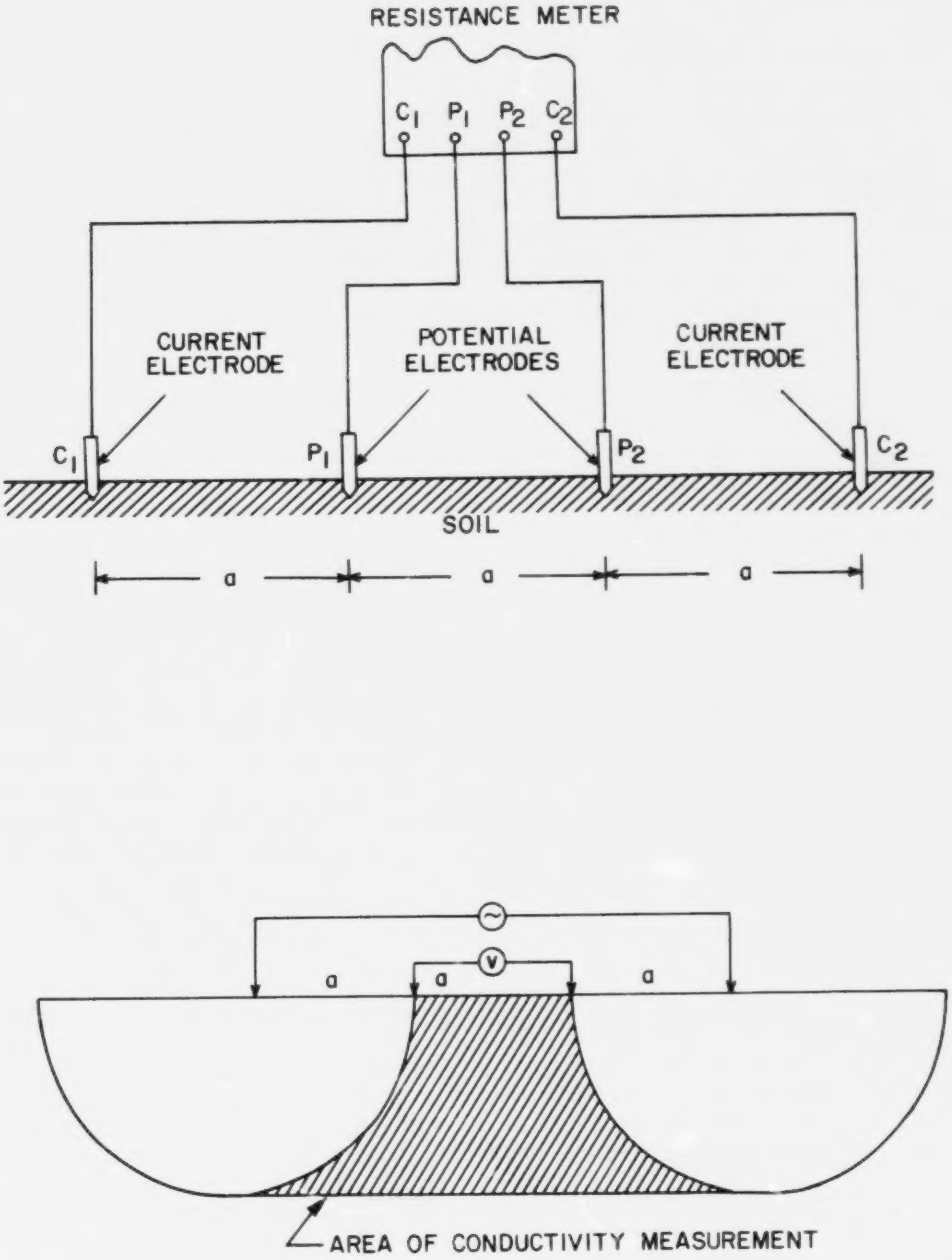

RESISTANCE METER
C1 P1 P2 C2
CURRENT ELECTRODE
POTENTIAL ELECTRODES
CURRENT ELECTRODE
C1
P1
P2
C2
SOIL
a
a
a
~
V
a
a
a
AREA OF CONDUCTIVITY MEASUREMENT

Figure 30. Wenner four-probe array and associated area of conductivity measurement (after Rhoades and Halvorson, 1977).

f_t = a factor to adjust the reading of a reference temperature of 25°C.

EC_a is reported in mmhos/cm.

In the absence of layering, the depth of penetration of the electrical current is about one-third at the outer electrode spacing, y (Rhoades, 1979a). Similarly, the volume of soil measured by a given "a" spacing in a single EC_a determination is about $\pi y^3/9$. To increase the volume of soil scanned during a salinity survey, it is thus a matter of increasing the "a" spacing of the electrodes. The limitation on "a" spacing, and thus depth, is the degree of uniformity of texture within the subsoil. A depth of 4 feet appears to be a reasonable lower limit. Figure 30 depicts the theoretical mass of the soil scanned by the method.

Rhoades and Halvorson (1977) successfully used the four-probe method to determine the average bulk soil salinity in soils of the Northern Great Plains. The four-probe method was also used by Rhoades to detect the presence of a saline shallow water table by varying the "a" spacing. In two soils examined by this method, water tables were detected at depths of about 1 meter (3.3 feet) and 3 meters (9.8 feet). The conductance of the soil solution in discrete depth intervals can also be obtained by varying the "a" spacing and using a difference equation presented by Rhoades and Halvorson (1977). They used this method to delineate salt-affected regions in incremental soil depths in Montana.

According to Rhoades (1979a), one of the major advantages of the four-probe method is that because of the high degree of correlation between EC_e and EC_a, the method facilitates obtaining representative values of the salinity of typically heterogeneous soils without excessive expenditures of time and money. Other advantages cited by Rhoades (1979b) are: larger volumes of the salinity of soil are scanned than when soil samples, suction cups, or salinity sensors are used; and because the method is fast and simple, it is particularly well suited for routine salinity monitoring and mapping. A disadvantage is that the accuracy of the method decreases when soil layering becomes pronounced.

The EC-probe--Rhoades and van Schilfgaarde (1976) developed an "EC-probe" to detect changes in soil salinity in discrete depth intervals within stratified soils. Figure 31 shows construction features of the unit. The probe utilizes the Wenner array of four electrodes with a fixed "a" spacing of 2.6 cm (1.02 inches). Salinity can thus be assessed in 15-cm (5.9-inch) depth intervals. The corresponding soil volume scanned with this electrode setting is about 90 cm^3 (5.5 in^3). The electrodes in the unit designed by Rhoades and van Schilfgaarde (1976) were constructed of brass embedded within a lucite housing. The probe is attached to a shaft with a handle. Wire leads from the electrodes are brought to the surface through the hollow shaft and handle. The outer two electrodes are attached to a current source and the inner two electrodes are attached to a resistance meter.

In operation, a soil cavity is augered to the depth of interest using a soil auger or sampler of about the same diameter as the probe. The probe is

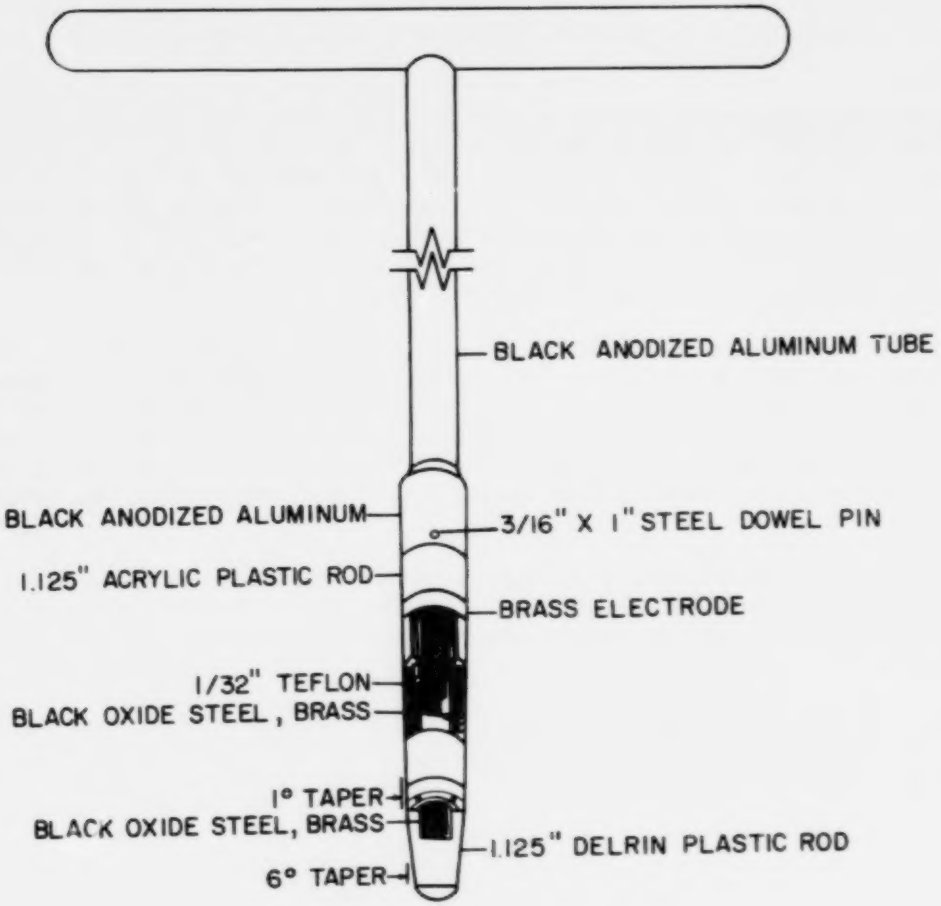

BLACK ANODIZED ALUMINUM TUBE
BLACK ANODIZED ALUMINUM
3/16" X 1" STEEL DOWEL PIN
1.125" ACRYLIC PLASTIC ROD
BRASS ELECTRODE
1/32" TEFLON
BLACK OXIDE STEEL, BRASS
1° TAPER
BLACK OXIDE STEEL, BRASS
1.125" DELRIN PLASTIC ROD
6° TAPER

Figure 31. Cross section of EC probe (after Rhoades and Halvorson, 1977).

placed within the cavity. An electric current is applied to the outer electrodes and the potential drop between the inner electrodes is measured via a resistance meter. The resistance value is converted to conductance by means of the cell constant for the probe. The cell constant is determined by submerging the unit in a container filled with solutions of known conductivity. Individual EC_a versus EC_e calibrations must be obtained for each of the soil strata. During operation of the probe in the field, it is important to measure salinity at a uniform water content value, such as that at field capacity.

Frequently during salinity studies, it is advantageous to obtain periodic measurements at the same location. For such cases, Rhoades (1979c) designed an inexpensive probe which can be left in place within a cavity. "... Implanted probes offer certain advantages, like avoiding the need for making new access holes, remaining in the same position with time, minimizing the complications arising from several access holes being made in the sampling area...." Ostensibly, a number of probes could be clustered within a common hole.

Four-electrode conductivity cell--The four-electrode conductivity cell is described in detail by Rhoades et al. (1977). Basically, the cell is constructed as follows. Undisturbed soil cores are obtained using a soil-core sampler with lucite column sections as core inserts. The lucite sections are removed from the sampler and segmented to form individual cells. Electrodes are inserted into threaded holes in the cell walls. The EC_a of the soil is subsequently measured using a resistivity meter.

In the unit described by Rhoades et al. (1977), light stainless steel electrodes are inserted into the soil surface at 45-degree intervals around a circumference to a fixed depth of 3 mm (0.118 inch). Consequently, any four neighboring electrodes could be regarded as a Wenner array, the outer two electrodes being current electrodes and the inner two electrodes being potential electrodes. By rotating the connectors, eight independent EC_a measurements can be made.

A cell constant must be obtained to permit converting resistance values to conductivity values. In practice, cell constants are obtained by filling the units with solutions of known conductance and measuring the cell resistance.

Methods for obtaining EC_e versus EC_a calibrations--Field measurements of EC_a are converted into corresponding EC_e values via calibration relationships. Rhoades and Halvorson (1977) discussed three calibration methods in detail. In particular, measurements of conductivity are obtained using the Wenner four-electrode method, the EC-probe, or the four-electrode cell. Soil samples are then obtained within the volume of soil measured by the selected method and laboratory measurements are made to determine the conductivity of the saturated extract, EC_e. A number of soil samples of differing salinity must be measured in order to obtain a range of EC_e versus EC_a values. In addition, the field EC_a measurements should be obtained at a uniform water content, such as that at field capacity.

Earth Resistivity Profiles for Delineating Pollution Plumes--

The electrical resistivity/conductivity method described above is useful for assessing salinity in the soil zone. The method has also been successfully used to monitor salinity in perched groundwater regions of the vadose zone. Again, the technique is based on the observation that the resistivity of solute in the pore space of the vadose zone offers less resistivity to current flow than dry rocks. By the same token, a highly salinized groundwater plume offers less resistivity to an applied current than less saline native groundwater. Consequently, by obtaining a series of resistivity-depth profiles in a grid within a waste disposal site, the lateral extent, and possibly the depth, of the plume could be delineated (Fenn et al., 1977).

Cartwright and his associates observed pollution plumes emanating from landfills in Illinois using earth resistivity surveying (Cartwright and

McComas, 1968; Cartwright and Sherman, 1972). Warner (1969) discussed the use of the method as a preliminary approach for delineating zones of polluted groundwater.

The basic approach used for earth resistivity surveys is the same as that for soil salinity surveys. That is, the Wenner array is used with increasing "a" spacings in order to observe resistivity changes with depth. According to Fenn et al. (1977), it is more appropriate to refer to "apparent resistivity" rather than "resistivity" alone. This term accounts for subsurface heterogeneity and is defined as "... the weighted average of the actual resistivities of the individual subsurface materials or strata within the depth of penetration at the resistivity measurement" (Fenn et al., 1977).

Two methods are used to interpret the field resistivity data: theoretical or empirical (Fenn et al., 1977). For the theoretical approach, the field resistivity values are plotted. "... The resulting curve (is) compared with sets of master curves developed for numbers of resistivity layers with definite ratios of resistivity and thickness. By this method, the value of resistivity for each geologic unit as well as its thickness and depth can be determined." The empirical method entails relating the field resistivity data to geological information, i.e., the properties of layered sediments. The success of the empirical approach thus depends greatly on the availability of geological information, such as texture and the vertical and areal distribution of the layered sediments, and the chemical quality of perched groundwater.

Stollar and Roux (1975) indicated that the overall success of the resistivity method in delineating plumes depends on the interrelationships of four factors: (1) the contrast between the conductivities of polluted and natural groundwater, (2) the depth to the top of the polluted groundwater body, (3) the thickness of the polluted groundwater body, and (4) lateral variations in surficial geology. Results of four resistivity surveys were presented by Stollar and Roux to demonstrate the importance of these interrelationships. Control data were obtained using geological information from existing wells or by installing test wells. The case studies included pollution plumes from industrial waste disposal sites and landfills. The depths to polluted groundwater ranged from 5 feet to 60 feet below sand surface. The method was successful in delineating the plume in all except one case. In the one unsuccessful case, it was determined that the following adverse conditions existed: (1) man-made obstructions were extensive over large portions of the area, (2) the water table was relatively deep, (3) the difference in conductivity value of polluted and natural groundwater was minor, and (4) extreme lateral variations in geology existed above the contaminated plume.

Salinity Sensors--

Another indirect method for in-situ evaluation of soil salinity is the so-called "salinity sensor." The basis of these devices is the relationship between specific electrical conductance, EC_e, of soil solution and the total concentration of salts in solution.

As described by Richards (1966), the basic idea of the salinity sensor is that electrodes embedded in porous ceramic, forming a hydraulic continuum with soil water, can be used to directly measure the specific conductance of the soil solution. From suitable calibration relations, therefore, specific conductance values can be directly related to the total salt content. The unit described by Richards comprised a plate about 1-mm (0.039-inch) thick with platinum electrodes fired in-place on opposing faces. An important feature of this sensor is that the unit is spring loaded so that good contact with soil could be ensured.

Because of the strong dependency of specific conductance on temperature, it is important to accurately measure the temperature of the soil solution. Richards used a thermistor to provide temperature compensation in his unit.

Oster and Willardson (1971) reviewed problems arising from calibration of sensors. They also reported on field studies using the unit. Of particular importance was their observation that sensors should not be used at soil-water pressures less than -2 atmospheres. Also, they indicated that when sensors are placed in trenches at field sites, the permeability of the materials in the backfilled trench tends to be greater than in indigenous soil. During leaching trials, therefore, the salinity measured with the sensors tended to be lower than in adjacent soil. Differences were attributed to greater leaching in the trench. However, structural differences were also present, i.e., the pore sizes in the trench soil were probably of a different range than indigenous soil.

Rhoades (1979b) indicated that a principal advantage of using salinity sensors compared to soil sampling for assessing salinity is that readings are taken in the same location. Thus, by installing a network of sensors, a chronological record of in-situ salinity changes can be obtained from a large volume of soil. Rhoades also indicated that salinity sensors "... are simple, easily read, and sufficiently accurate for salinity monitoring purposes."

The U.S. Salinity Laboratory Staff (1977) discussed the use of a data acquisition system which automatically obtains electrical conductivity values from a number of salinity sensors. The units were installed during field studies on salt movement in irrigation return flow. By means of a computer program, the system activated stepping switches, read the sensors in sequence, adjusted the readings for soil temperature, and printed out the calculated EC values. The resultant data were transmitted by means of a teletype to Riverside, California.

Rhoades (1979b) compared the use of salinity sensors and four-probe units for assessing soil salinity. He concluded that four-probe units have the following six major advantages over the salinity sensor: (1) they are less subject to calibration change, (2) they are more durable and less costly, (3) they do not suffer from time lags in the response to changing salinity, (4) they can be sized to measure within different soil volumes, (5) they are more versatile, and (6) they are well suited for mapping and diagnosis, as well as monitoring.

Solids Sampling in the Vadose Zone

An important aspect of a monitoring program for the vadose zone underlying a waste disposal operation is to obtain samples of the solid materials throughout this region. Solids are defined as soils and underlying geologic materials. The rationale for solids sampling in the vadose zone was stated by Dunlap et al. (1977):

> Only by analysis of earth solids from the unsaturated zone underlying pollutant-releasing activities can those pollutants which are moving very slowly toward the water table because of sorption and/or physical impediment be detected and their rates of movement and degradation measured. Such pollutants, which probably include a major proportion of organics and microorganisms, are not likely to be detected in groundwater until the activities releasing them have been in operation for protracted periods. Because of their potential for long-term pollution of groundwater, it is imperative that the behavior of these pollutants in the subsurface be established at the earliest practicable time.

Dunlap et al. (1977) also listed reasons for sampling from the groundwater zone ("zone of saturation"). These reasons apply equally well for sampling from perched groundwater:

> Analyses of organic pollutants in solid samples from the zone of saturation are needed for a realistic evaluation of the total extent and probable longevity of organic pollution in an aquifer. Such analyses provide a measure of the quantity of pollutants which are sorbed on aquifer solids and which are in equilibrium with, and in essence serve as a reservoir for, pollutants in solution in the adjacent groundwater.
>
> Microbial populations which may be involved in the biological alteration of pollutants in subsurface formations are likely to be in such close association with subsurface solids that they will not be present in well waters in numbers which are quantitatively indicative of their presence in the formations; hence, analyses of subsurface solids are needed for accurate evaluation of such populations.
>
> Even when the best well construction and groundwater sampling procedures are used, it is difficult to completely eliminate the possibility that contaminating surface microbes may be present in groundwater samples. Solids taken from the interior of cores carefully obtained from the zone of saturation probably provide the most authentic samples of aquifer microorganisms that can be obtained.

In this section, the following items are discussed: (1) soils sampling methods for inorganic chemicals, organic constituents, and microorganisms; (2) sampling of solids in the lower vadose zone; (3) field analyses; (4) field and laboratory handling of samples; and (5) advantages and disadvantages of solids sampling methods.

Soils Sampling--

Soil sampling methods used by soil scientists and irrigation and drainage engineers to evaluate the physical properties of soils are also suitable for sampling to determine inorganic chemical constituents. Particular methods which have been used include the following types of hand augers: screw-type augers, post-hole augers, barrel augers, and dutch augers (Donnan, 1957). Another common type of sampler is the split-spoon sampler, which is a barrel-type auger, one side of which pivots on a hinge. Tube-type samplers have been used extensively to obtain soil cores. The Veihmeyer sampler is an example of such a unit. The end of the sampler is bevelled and sharpened to facilitate insertion into the soil. A drive hammer is used to force the tube to the desired depth. An intact core is removed from the tube. Such a core may show the distinct breaks in soil layering.

A problem associated with using augers and tube-type samplers is that if the soil is very dry, the sample tends to fall out of the unit when it is being withdrawn. "Core catchers" could be used to overcome this problem (Thomas, personal communication, 1979). Another problem is that if the soil is wet, some chemical interactions may occur between the soil water and the metal parts of the sampler. Thus, trace metals could be introduced into the sample.

The problem of contamination between a soil sample and sampling device is of particular importance when sampling for organic chemicals and microorganisms. Bordner, Winter, and Scarpino (1978) recommended the following procedure for obtaining soil samples for microbial analysis:

> ... scrape the top one inch of soil from a square foot area using a sterile scoop or spoon. If a subsurface sample is desired, use a sterile scoop or spatula to remove the top surface of one inch or more from a one-foot square area. Use a second scoop or spoon to take the sample. Place samplings in a sterile one quart screw-cap bottle until it is full. Depending on the amount of moisture, a one-quart bottle holds 300-800 grams of soil. Label and tag the bottle carefully and store at 4°C until analyzed.

Techniques for sampling organic and microbial constituents have been examined by The Groundwater Research Branch of the Robert S. Kerr Environmental Research Laboratory, as reported by Dunlap et al. (1977). One method used with success comprises an auger and dry-tube coring procedure. The method entails augering a hole to the top of the desired sampling depth and forcing a "dry-tube" core sampler into the sampling region. The core sampler used by Dunlap et al. (1977) was a steel tube about 18 inches long and 3 inches in diameter. The core barrel is fitted with a steel drive shoe which is of slightly smaller inside diameter than the diameter of the barrel. This arrangement facilitates removing the core. As pointed out by Dunlap et al. (1977), contamination problems are minimized by the auger/dry-tube coring technique primarily because drilling fluid is not used. It would appear that the method could be modified to include a lucite plastic insert in the core barrel to minimize contaminating soil samples obtained for trace metal analysis.

Sampling from the Lower Vadose Zone--

The methods discussed above for obtaining soil samples could, with difficulty, be used to sample from the lower vadose zone. For example, Fenn et al. (1977) indicated that post-hole augers could be used to sample to depths down to 80 feet in certain material by using a tripod and pulley. A more convenient faster approach is to employ engine-driven augers and drilling equipment. Among the power-driven sampling units available are continuous-flight spiral augers (which could be hollow-stemmed), core samplers, bucket augers, cable-tool drill rigs, and rotary drill rigs. The spiral-type units are not particularly suitable for sampling because it is not always possible to distinguish the sampling depth. By using the hollow-stem auger, however, it is possible to insert a core sampler inside the auger and sample from discrete depths. Soil sampling machines are available commercially for obtaining either auger samples or core samples by attaching appropriate tools. These machines are mounted on trailers with independent engines. Alternatively, they may be purchased for mounting on pick-up trucks and driven by power takeoffs. Hydraulic controls facilitate drilling or coring operations. Travelling Kelley bars permit sampling at depths greater than 25 feet (courtesy, Giddings Machine Company).

A bucket auger consists of a large-diameter bucket fitted with cutting blades (Fenn et al., 1977). The bucket is rotated in the hole until filled. When brought to the surface, the bucket is dumped and a sample can be easily obtained. Alternatively, a core is obtained prior to dumping. Rible et al. (1976) used this method to sample for nitrate distribution in the vadoze zone underlying irrigated fields in California.

Drill rigs could be used for sampling purposes at the same time that test wells and monitor wells are being installed at the waste disposal site. The cable-tool, or percussion, technique entails alternately raising and dropping a heavy string of tools in the cavity. The drill bit crushes large-diameter material. Water is commonly added to the borehole and, after removing the tool string, a bailer is inserted. The fine cuttings are removed by the bailer. Casing is driven in as drilling progresses. An obvious disadvantage of this method is that the water added during drilling changes the chemistry of the native pore water. This problem may be obviated somewhat by analyzing the water and adjusting the analyses obtained from solids samples.

The hydraulic rotary method utilizes a rotating bit to excavate a borehole and a continuous flow of water to remove drill cutting. Drillers' mud is added to the circulating water during drilling to hold open the hole. Obviously, solids samples would be badly contaminated by the mud/water mixture used with this method. Also, the precise depths at which cuttings are obtained are impossible to define.

In the air rotary method, air is used instead of water to bring the drill cuttings to the surface. Thus, the problem of contamination encountered with the regular hydraulic rotary drilling method is not a factor in air drilling. There may be difficulty, however, in defining the sampling depth precisely, i.e., particle-size segregation may occur because of density differences.

The core sampling method of Dunlap et al. (1977) has been used to obtain solids samples for analyses of organic chemicals and microorganisms in depths to about 25 feet. In addition to dry samples, solids have also been obtained from saturated regions. Problems developed, however, when sampling loose saturated material. An alternative approach to overcome such problems which was considered by Dunlap et al. was to use a hollow-stem auger in lieu of a regular auger. Dunlap et al. also described a piston-type sampler for collecting solids for organic analyses when fluid is used in the drilling process. To date, the sampler has not been extensively field tested.

Field Analysis of Pore Water--

Whenever possible, pore water should be extracted from solid samples in the field and analyzed immediately for unstable constituents such as temperature, pH, eH, and EC (Mooij and Rovers, 1976). When the soils are unsaturated, special techniques are required to obtain samples of the pore water. Fenn et al. (1977) reviewed a number of alternative methods for extracting pore water from core samples, including a method by Lusczynski (1961) which uses CO_2 under pressure to displace the fluid. Another method described by Fenn et al. is a "hydraulic squeezer," which uses a pressurized piston to force pore water into a syringe.

Instead of using pore water directly, it is often more convenient to prepare saturated pastes in the field and extract water for specific conductance measurements. Rhoades and Halvorson (1977) described a convenient extraction unit for field use. Saturated paste is transferred into a small-diameter filter funnel made of plastic with a filter pipe in place. Vacuum is provided by a hand pump and the sample is collected in a reservoir. The EC of the sample is determined via a portable EC bridge.

Field and Laboratory Preparation of Samples--

Dry solids samples collected for chemical analyses of pore water in the laboratory should be stored in paper bags. If the samples are wet, it will be necessary to store the samples in plastic bags. Depending on the intended analyses, the samples are air-dried in the laboratory. If the analyses are intended for the nitrogen series, the atmosphere in the handling room should be free of ammonia.

Samples collected for determining microorganisms and organic pollutants using the "dry-tube" core sampler require special handling. Dunlap et al. (1977) described the procedure in detail. Briefly, the core is pressed from the barrel of the sampler and the first 5 to 8 cm (2 to 3.1 inches) of solids is removed for analysis of physical/chemical parameters. In order to obviate microbial contamination of the core sample by contact with the walls of the sampler, a subsample is removed from the center of the core (see Figure 32).

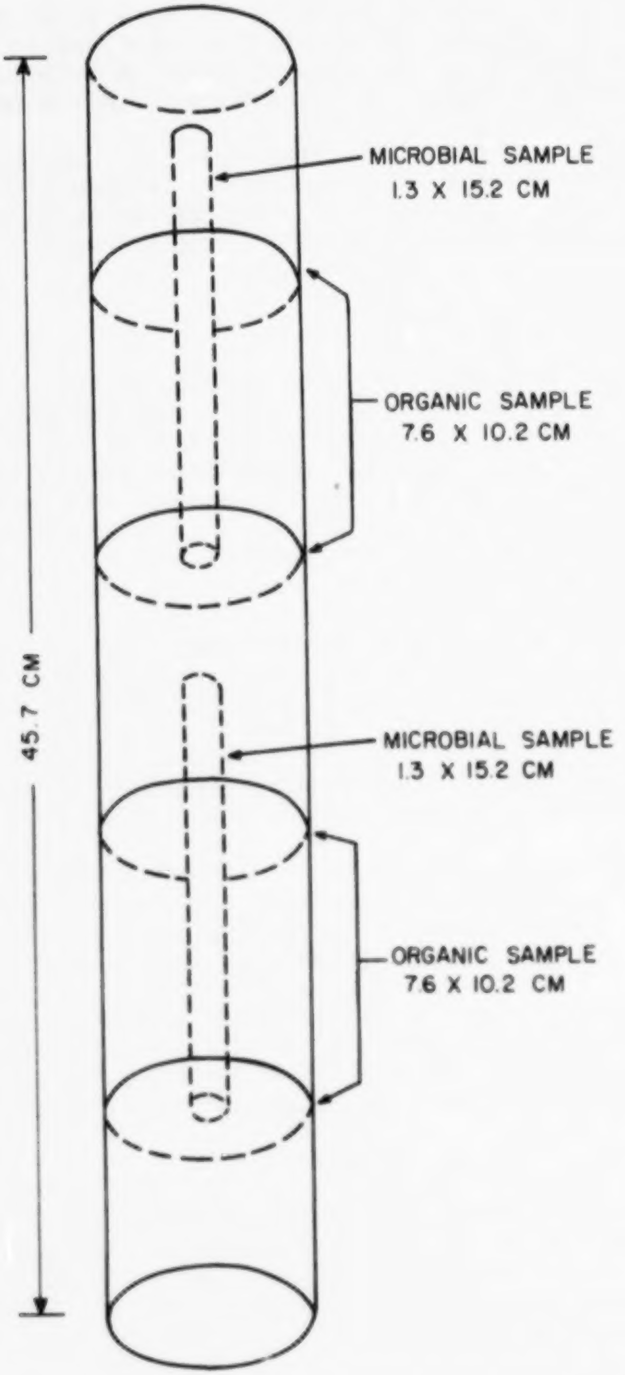

MICROBIAL SAMPLE
1.3 X 15.2 CM
ORGANIC SAMPLE
7.6 X 10.2 CM
45.7 CM
MICROBIAL SAMPLE
1.3 X 15.2 CM
ORGANIC SAMPLE
7.6 X 10.2 CM

Figure 32. Subsamples from a "dry tube" core sampler (after Dunlap et al., 1977).

This is effected by inserting a sterile stainless steel tube into the center of the core. The subsample is extracted using a sterile rod and placed in containers. Subsamples for analyses of adenoside triphosphate (ATP) are obtained in a similar manner, i.e., subsampling from the center of the core.

Two 5-cm (2-inch) long subsamples for organic analyses are obtained after removing the central cores. The subsamples are dropped into aluminum baking pans and covered with aluminum foil. The pan and subsamples are then placed into an insulated polystyrene box with liquid nitrogen to quick-freeze the sample.

Problems With Solids Sampling--

An important limitation of field sampling of solids is that an inordinate number of samples may be required to ensure that the measured mean value of a particular parameter falls within a given range of a true value. A principal reason for the problem is that variations arise because of the natural spatial variability of soil properties. Such variations were noted by Rible et al. (1976); Pratt, Warneke, and Nash (1976); and Lund and Pratt (1977) during studies on nitrate leaching in irrigated fields in California. Pratt, Warneke, and Nash (1976), for example, showed that five to 21 sampling sites were needed (eight samples per site) from four fields ranging from 62 to 223 acres to obtain measured mean nitrate concentrations that were within 30 percent of the true mean at the 95-percent confidence limit. They also estimated that it would have been necessary to sample 23 to 161 sites per field to be within 10 percent of the true mean.

In summarizing the utility of solids sampling for nitrate leaching from irrigated fields, Lund and Pratt (1977) stated:

> Monitoring nitrate leaching by soil sampling does not appear to be feasible for basin-wide studies. As a research tool to study the components of the nitrogen cycle it has proved useful, but we would not recommend it as a monitoring method.

Rhoades (1979b) described another problem with solids sampling: "Some changes in soil-water composition occur as soil is removed from its natural condition, dried, ground, sieved, extracted, etc., hence only relative changes in composition, not absolute, can be determined from soil sampling."

Solution Sampling in Unsaturated Media

Wells and open cavities cannot be used to collect solution flowing in the vadose zone under suction (negative pressures). The sampling devices for such unsaturated media are thus called suction samplers. Three types will be discussed: (1) ceramic-type samplers, (2) hollow fiber samplers, and (3) membrane filter samplers.

Ceramic-Type Samplers--

There are two types of samplers constructed from ceramic materials: the suction cup and the filter candle. Both types operate in the same manner.

Basically, ceramic-type samplers (also called suction "lysimeters") comprise the same type of ceramic cups used in tensiometers. When placed in the soil, the pores in these cups become an extension of the pore space of the soil. Consequently, the water content of the soil and cup become equilibrated at the existing soil-water pressure. By applying a vacuum to the interior of the cup such that the pressure is slightly less inside the cups than in the soil solution, flow occurs into the cup. By an appropriate technique, the sample is pumped to the surface, permitting laboratory determination of the quality of the soil solution in-situ. Although cups have limitations, at the present time they appear to be the best tool available for sampling unsaturated media, particularly in the field.

Suction cups may be subdivided into three categories: (1) vacuum operated soil-water samplers, (2) vacuum-pressure samplers, and (3) vacuum-pressure samplers with check valves. The soil-water samplers generally consist of a ceramic cup mounted on the end of a small-diameter PVC tube, similar to a tensiometer (see Figure 33). The upper end of the PVC tubing projects above the soil surface. A rubber stopper and outlet tubing are inserted

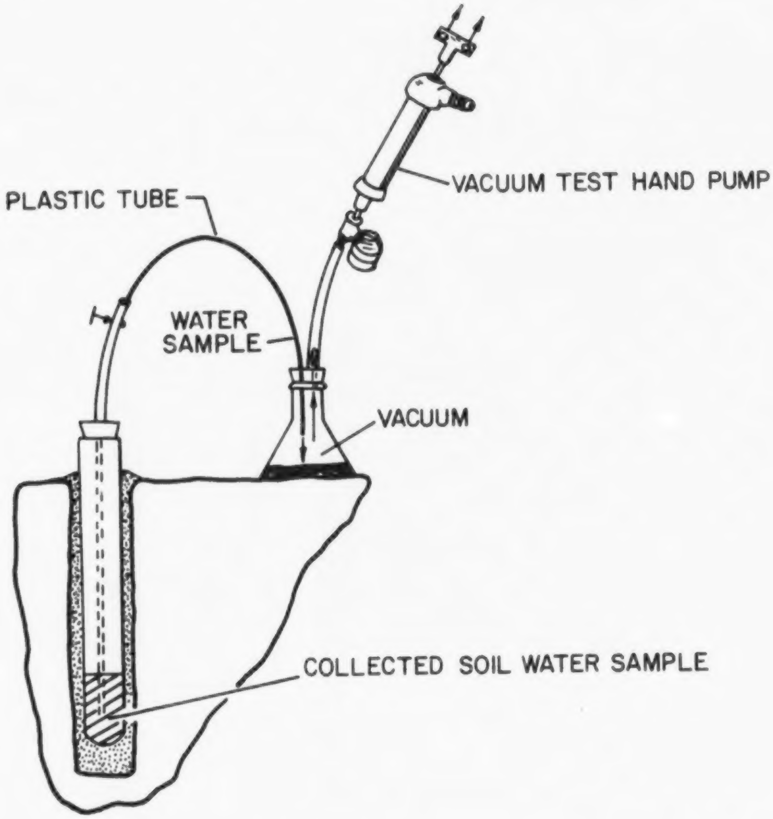


Figure 33. Soil-water sampler (courtesy, Soil Moisture Equipment Corporation, 1978).

into the upper end. Vacuum is applied to the system and soil water moves into the cup. To extract a sample, a small-diameter tube is inserted within the outlet tubing and extended to the base of the cup. The small-diameter tubing is connected to a sample-collection flask. A vacuum is applied via a hand vacuum-pressure pump and the sample is sucked into the collection flask. These units are generally used to sample to depths up to 6 feet from the land surface. Consequently, they are used primarily to monitor the near-surface movement of pollutants from land disposal facilities or from irrigation return flow.

To extract samples from depths greater than the suction lift of water, a second type of unit is available, the so-called vacuum-pressure lysimeter. These units were developed by Parizek and Lane (1970) for sampling the deep movement of pollutants from a land disposal project in Pennsylvania. The design of the Parizek and Lane sampler is shown on Figure 34. The body tube of the unit is about 2-feet long, holding about 1 liter (0.26 gallon) of sample.

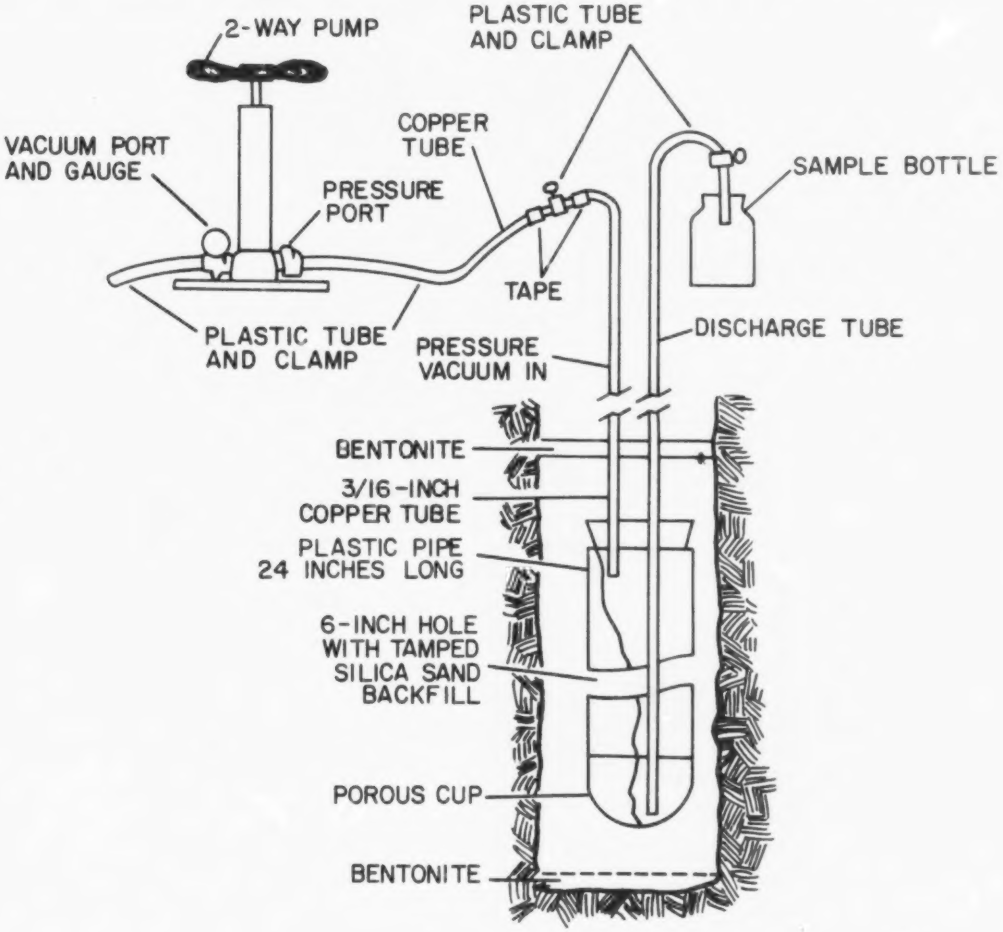

2-WAY PUMP
PLASTIC TUBE AND CLAMP
VACUUM PORT AND GAUGE
COPPER TUBE
SAMPLE BOTTLE
PRESSURE PORT
TAPE
PLASTIC TUBE AND CLAMP
PRESSURE VACUUM IN
DISCHARGE TUBE
BENTONITE
3/16-INCH COPPER TUBE
PLASTIC PIPE 24 INCHES LONG
6-INCH HOLE WITH TAMPED SILICA SAND BACKFILL
POROUS CUP
BENTONITE

Figure 34. Vacuum-pressure sampler (after Parizek and Lane, 1970).

Two copper lines are forced through a two-hole rubber stopper sealed into a body tube. One copper line extends to the base of the ceramic cup as shown and the other terminates a short distance below the rubber stopper. The larger line connects to a sample bottle and the shorter line connects to a vacuum-pressure pump. Care must be exercised to ensure that all lines and connections are sealed.

In operation, a vacuum is applied to the system (the longer tube to the sample bottle is clamped shut at this time). When sufficient time has been allowed for the unit to fill with solution, the vacuum is released and the clamp on the outlet line is opened. Air pressure is then applied to the system forcing sample into the collection flask. A basic problem with this unit is that when air pressure is applied, some of the solution in the cup may be forced back through the cup into the surrounding pore water system. Consequently, this type of pressure-vacuum system is recommended for depths only up to about 50 feet below land surface. In addition to the monitoring effort of Parizek and Lane, these units were used by Apgar and Langmuir (1971) to sample leachate movement in the vadose zone underlying a sanitary landfill.

Wood (1973) reported on a modified version of the design of Perizek and Lane. Wood's design is the third suction sampler discussed in this section. The design of Wood overcomes the main problem of the simple pressure-vacuum system; namely, that solution is forced out of the cup during application of pressure. A sketch of the sampler is shown on Figure 35. Basically, the cup ensemble is divided into lower and upper chambers. The two chambers are isolated except for a connecting tubing with a check valve. A sample delivery tube extends from the base of the upper chamber to the surface. This tube also contains a check valve. A second shorter tube terminating at the top of the sampler is used to deliver vacuum or pressure. In operation, when a vacuum is applied to the system, negative pressure extends to the cup through the open one-way check valve. The second check valve in the delivery tube is shut. The sample is delivered into the upper chamber, which is about 1 liter (0.26 gallon) in capacity. To deliver the sample to the surface, the vacuum is released and pressure (generally of nitrogen gas) is applied to the shorter tube. The one-way valve to the cup is shut and the one-way valve in the delivery tube is opened. Sample is then forced to the surface. High pressures can be applied with this unit without danger of damaging the cup. Consequently, this sampler can be used to depths of about 150 feet below land surface (Soil Moisture Equipment Corporation, 1978). Wood and Signor (1975) used this sampler to examine geochemical changes in water during flow in the vadose zone underlying recharge basins in Texas.

A sampling unit employing a filter candle was described by Duke and Haise (1973). The unit, described as a "vacuum extractor," is installed below plant roots. Figure 36 shows an illustrative installation. The unit consists of a galvanized sheet metal trough open at the top. A porous ceramic candle (12 inches long and 1.27 inches in diameter) is placed into the base of the trough. A plastic pipe sealed into one end of the candle is connected to a sample bottle located in a nearby manhole or trench. A small-diameter tube attached to the other end of the candle is used to rewet the candle as necessary. The trough is filled with soil and placed within a horizontal cavity of the same dimensions as the trough. The trough and enclosed filter candle are

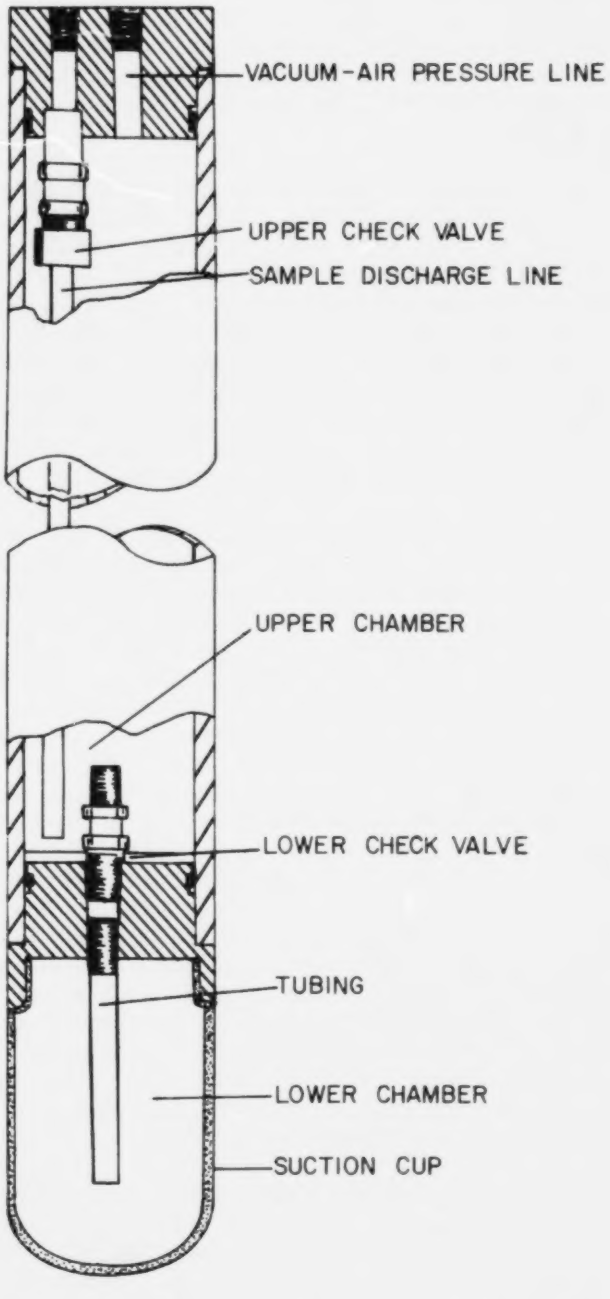

VACUUM-AIR PRESSURE LINE
UPPER CHECK VALVE
SAMPLE DISCHARGE LINE
UPPER CHAMBER
LOWER CHECK VALVE
TUBING
LOWER CHAMBER
SUCTION CUP

Figure 35. "Hi/Pressure-vacuum soil-water sampler" (courtesy, Soil Moisture Equipment Corporation, 1978).

pressed up against the soil via an air pillow or mechanical jack. In operation, vacuum is applied to the system such that soil water is induced to flow into the trough and candle at the same rate as in the surrounding soil. The amount of vacuum is determined from tensiometers. Hoffman et al. (1978) used this type of sampler to collect samples of irrigation water leaching beneath the roots of orange trees during return flow studies at Tacna, Arizona (see Figure 36).

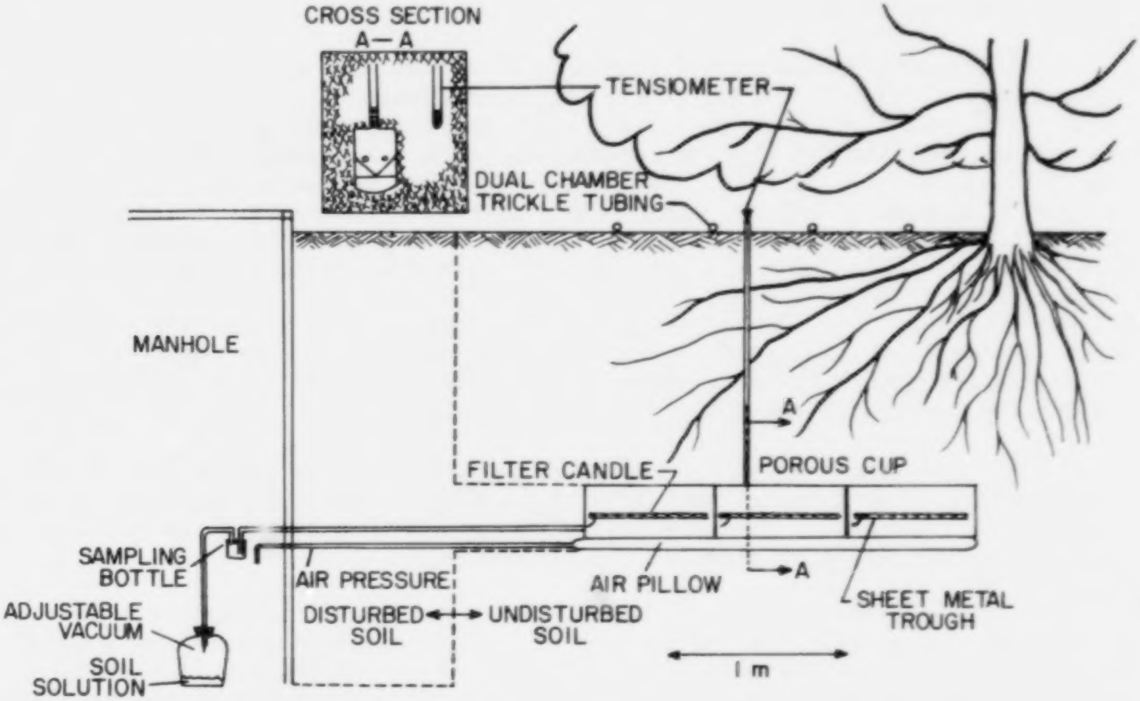

CROSS SECTION
A—A
TENSIOMETER
DUAL CHAMBER
TRICKLE TUBING
MANHOLE
A
FILTER CANDLE
POROUS CUP
SAMPLING
BOTTLE
AIR PRESSURE
AIR PILLOW
A
SHEET METAL
TROUGH
ADJUSTABLE
VACUUM
DISTURBED
SOIL
UNDISTURBED
SOIL
SOIL
SOLUTION
1 m

Figure 36. Facilities for sampling irrigation return flow via filter candles, for research project at Tacna, Arizona (after Hoffman et al., 1978).

Sampler preparation, installation, and operation--Laboratory studies on ceramic cups by Wolff (1967) showed that new cups yield several milligrams per liter of Ca, Mg, Na, HCO_3, and SiO_2. Consequently, it is recommended that cups be leached prior to installation. In his studies, Wood (1973) cleaned new cups as follows: "... the cups were cleaned by letting approximately 1 liter of 8N HCl seep through them. This acid treatment was followed by allowing 15 to 20 liters of distilled water to seep through and rinse the cups thoroughly. The cups are adequately rinsed when there is less than a 2% difference between the specific conductance of the distilled water input and the output from the cup."

Installation of the shallow vacuum-type samplers is similar to the installation of tensiometers. A cavity is first augered to the desired depth. Soil from the base of the hole is wetted to form a slurry (Soil Moisture

Equipment Corporation, 1978). The next step is to place the sampler into the hole, forcing the cup into the slurry. The entire cup should be covered by the slurry. The hole is then backfilled with soil, tamping to ensure that side leakage is prevented. An alternative installation method entails placing bentonite clay at the base of the open hole and placing a small amount of 200-mesh silica sand above the bentonite. The sampler is placed in the hole and a 6-inch layer of sand is added above the cup. Bentonite clay is then placed on top of the sand as a seal against side leakage of water. Finally, the hole is filled with native soil.

The pressure-vacuum type lysimeters, either with or without the check valves, may be installed singly in a cavity or clustered with several units in a cavity. Installation of the single units entails using one of the methods discussed above for the shallow vacuum-type units. An example of a cluster design is shown on Figure 37. In the illustration, two units are installed in a hole. The base of the hole should be sealed with bentonite. A layer of silica sand is then placed on top of the bentonite plug. The lowermost cup is placed on the silica sand and subsequently surrounded by more sand. Native soil is backfilled in the cavity to an elevation near that of the next sampler. Bentonite is added as a plug. Silica sand is then added, the second unit is installed, and backfilling proceeds as above. In some cases, it may be a good precaution to add a layer of bentonite above the final silica sand layer to minimize the possibility of side leakage.

The above-ground assembly comprises a pressure-vacuum system and collection bottles. For a system with several samplers, a manifold assembly is advantageous. Generally, pressure should be added with a gas such as nitrogen to preclude oxidation of chemical constituents.

For the shallow soil solution samplers, vacuum could be applied either as a constant vacuum or a falling vacuum. A constant vacuum is maintained using a vacuum pump with a preset vacuum. In the falling vacuum type, an initial vacuum is applied to the system and vacuum gradually decreases as the sample is drawn into the cups (Hansen and Harris, 1975). Problems may arise in the operation of the constant vacuum system because of differences in sample collection rate. In particular, some units fill more rapidly than others such that solution may overflow into adjoining collection flasks. Thus, cross-contamination may occur. Chow (1977) presented a design of a mercury-pressure control device to shut off the vacuum to a flask when filled to a preset level.

Vacuum extractor assemblies comprising sheet metal troughs and filter candles could be installed by a technique used by the U.S. Salinity Laboratory Staff (1977) during their irrigation return flow studies at Tacna, Arizona. In particular, the extractors were installed in a rectangular tunnel formed by first augering a horizontal hole at the desired depth. Subsequently, a rectangular shaper was forced into the tunnel. The metal troughs containing filter candles were filled with soil augered from the hole. The extractor assembly was then forced against the ceiling of the tunnel by inflating a Butyl rubber air pillow. Two tensiometers were installed over each extractor to facilitate determining the vacuum to the filter candles.

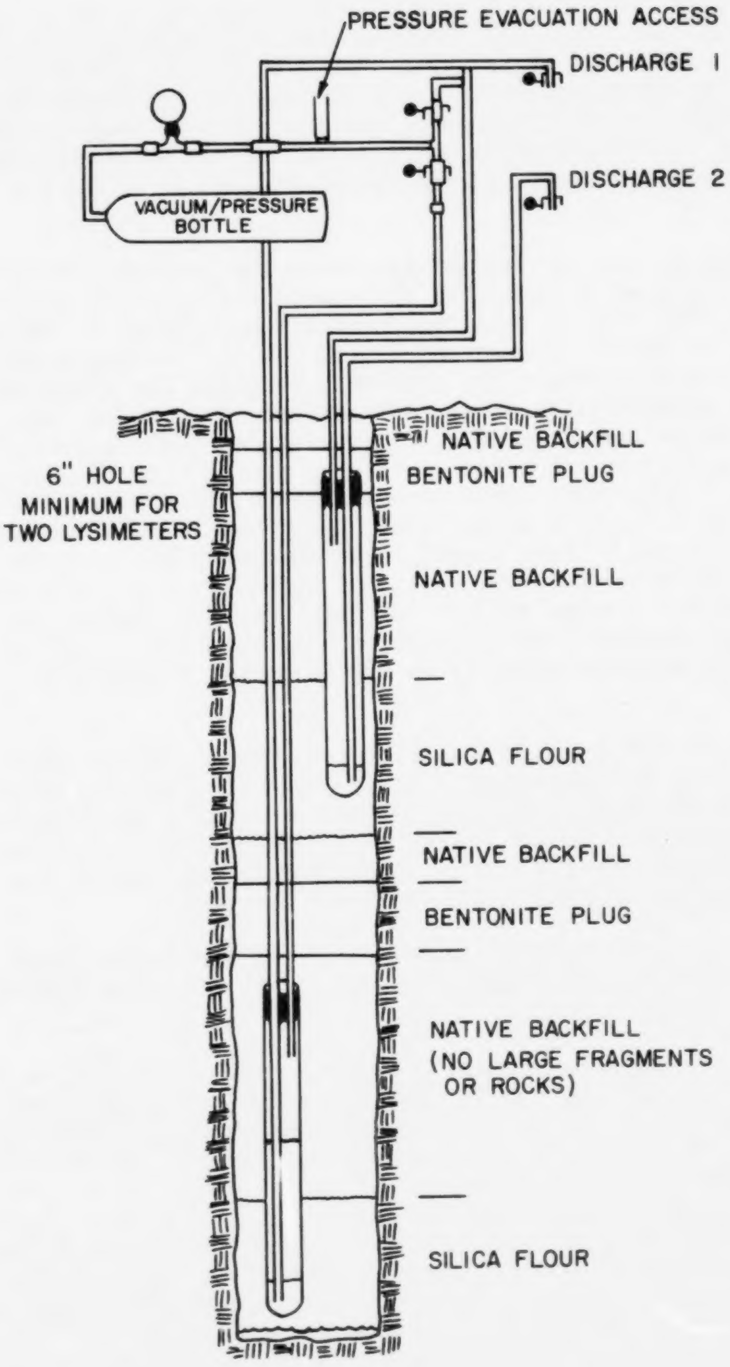

PRESSURE EVACUATION ACCESS
DISCHARGE 1
DISCHARGE 2
VACUUM/PRESSURE BOTTLE
NATIVE BACKFILL
6" HOLE MINIMUM FOR TWO LYSIMETERS
BENTONITE PLUG
NATIVE BACKFILL
SILICA FLOUR
NATIVE BACKFILL
BENTONITE PLUG
NATIVE BACKFILL (NO LARGE FRAGMENTS OR ROCKS)
SILICA FLOUR

Figure 37. Clustered suction cup lysimeters in a common borehole (after Hounslow et al., 1978).

Problems with ceramic-type samplers--Although the ceramic-type suction samplers are the best available technique for obtaining pore water samples in-situ, certain problems in their usage should be noted. One problem is that samples cannot be obtained over the entire range of soil-water pressures. That is, a ceramic unit is capable of sampling only to negative soil-water pressures equal to the air-entry value of the unit. Generally, this is about -1.0 bar, although units with air-entry values down to -2.0 bars are available.

In addition to the limitation imposed by the air-entry value of ceramic samplers, the representativeness of the sample collected is also questionable. In other words, how closely does the chemical composition of the sample represent that of pore water? Hansen and Harris (1975), in controlled laboratory and field tests using ceramic cup samplers, examined the effect of five factors on sample concentration during extraction: cup intake rate, leaching, diffusion, sorption, and screening. Screening, or "salt sieving," refers to a decrease in overall concentration of solutes during flow through a membrane. Leaching and diffusion were found to be of minor importance. Using phosporus and nitrate-N as indicators of sorption, they found that nitrate-N was not sorbed by ceramic cups. This result is in agreement with earlier studies by Wagner (1962) who noted that NO_3-N was not sorbed by cups, although sorption of NH_4-N did occur. Hansen and Harris (1975) found evidence that P was strongly sorbed. However, they concluded that sorption is serious only when "... the ceramic sorptive capacity is significantly greater than that of the adjacent soil."

The intake rate had a profound effect on sample concentration in a system with varying solute concentration. Thus, fast-rate samplers collected the largest fraction of their sample at the beginning of a sampling interval. Inasmuch as the concentration of solute in the pore water system was changing rapidly, the concentration of sample collected by the sampler was not truly representative of water draining to the water table.

The fact that ceramic-type samplers extract pore water under low suctions also affects the representativeness of the sample. England (1974) described two of the causative factors:

> First, numerous experimental and theoretical studies have clearly shown that the concentration and the composition of the soil solution are not homogeneous throughout its mass. Cations, in particular, vary widely in degree of dissociation from the surface of electronegative colloidal soil particles. Thus, water drained from large pores at low suctions may have a chemical 'quality' that is very different from that extracted from micropores. A point source of suction, such as the porous cup, samples roughly a sphere, draining different-sized pores as functions of distance from the point, the amount of applied suction, the hydraulic conductivity of the medium, and the soil-water content.
>
> Also, the concentrations of various ions in a soil solution generally do not vary inversely with the soil-water content.

> Reitemeier ... and many others have presented evidence that the total dissolved quantities of some ions increase on dilution while concurrently, those of other ions may decrease.

In light of these effects, Hansen and Harris (1975) concluded that each group of pore sizes has a unique volume of drainable water, ion concentration, and drainage-rate curve. They concluded that in order to collect a soil-water sample with an ion concentration representative of that draining to the water table, the rate of sample collection (and thus the incremental volumes) should correspond to the pore water drainage rate.

The problem of representativeness also arises when ceramic units are used for solution sampling in highly structured media. As noted by Shaffer, Fritton, and Baker (1979), the bulk of the flow occurs through the interped cracks. Consequently, units which sample pore water from within soil blocks do not sample from the bulk of the water passing into the lower vadose zone. In addition to well-structured soils, this physical situation could occur within spoil piles of coal strip-mined areas.

Field studies by Biggar and Nielsen (1976) illustrated a limitation of point samplers such as ceramic samplers (and fiber type samplers) for estimating the mass flux of solute beneath a given soil depth (mass flux of solute is the mass of solute crossing a unit area per unit time). In particular, because of the spatial variability of soil properties, the soil water velocity, and thus soil water flux, may not be normally distributed. In fact, for their test area, Biggar and Nielsen found velocity to be logarithmically normally distributed. Consequently, substantial errors could be made in estimating the mass flux of solute beneath a given soil depth by multiplying average values of the flux of water by average values of the concentration of the soil solution. "Hence, we expect point measurement such as the discharge from drain tiles, solution samples from suction probes and piezometer wells, and excavated soil samples to provide good indications of relative changes in the amount of solute sufficiently precise to ascertain the amounts of fertilizers or other solutes leached beyond the recall of roots. Only in case of a thorough analysis of the frequency distribution of such measurements would quantitative results be assured."

Cellulose-Acetate Hollow Fiber Samplers--

Jackson, Brinkley, and Bondietti (1976) described a suction sampler constructed of cellulose-acetate hollow fibers. These semipermeable fibers have been used for dialysis of aqueous solutions, functioning as molecular sieves. Soil column studies using a bundle of fibers to extract soil solution showed that the fibers were sufficiently permeable to permit rapid extraction of solution for analysis. Soil solution was extracted at soil-water contents ranging from 50 to 20 percent.

Levin and Jackson (1977) compared ceramic cup samplers and hollow fiber samplers for collecting soil solution samples from intact soil cores. Their conclusion was: "... porous cup lysimeters and hollow fibers are viable extraction devices for obtaining soil solution samples for determining EC, Ca, Mg, and PO_4-P. Their suitability for NO_3-N is questionnable." They also concluded that hollow fiber samplers are more suited to laboratory studies, where ceramic samplers are more useful for field sampling.

Membrane Filter Samplers--

Stevenson (1978) presented the design of a suction sampler using a membrane filter and a glass fiber prefilter mounted in a "Swinnex" type filter holder. The construction of the unit is shown on Figure 38. The membrane filters are composed of polycarbonate of cellulose-acetate. The "Swinnex" filter holders are manufactured by the Millipore Corporation for filtration of fluids delivered by syringe. A flexible tube is attached to the filter holder to permit applying a vacuum to the system and for delivering the sample to a bottle.

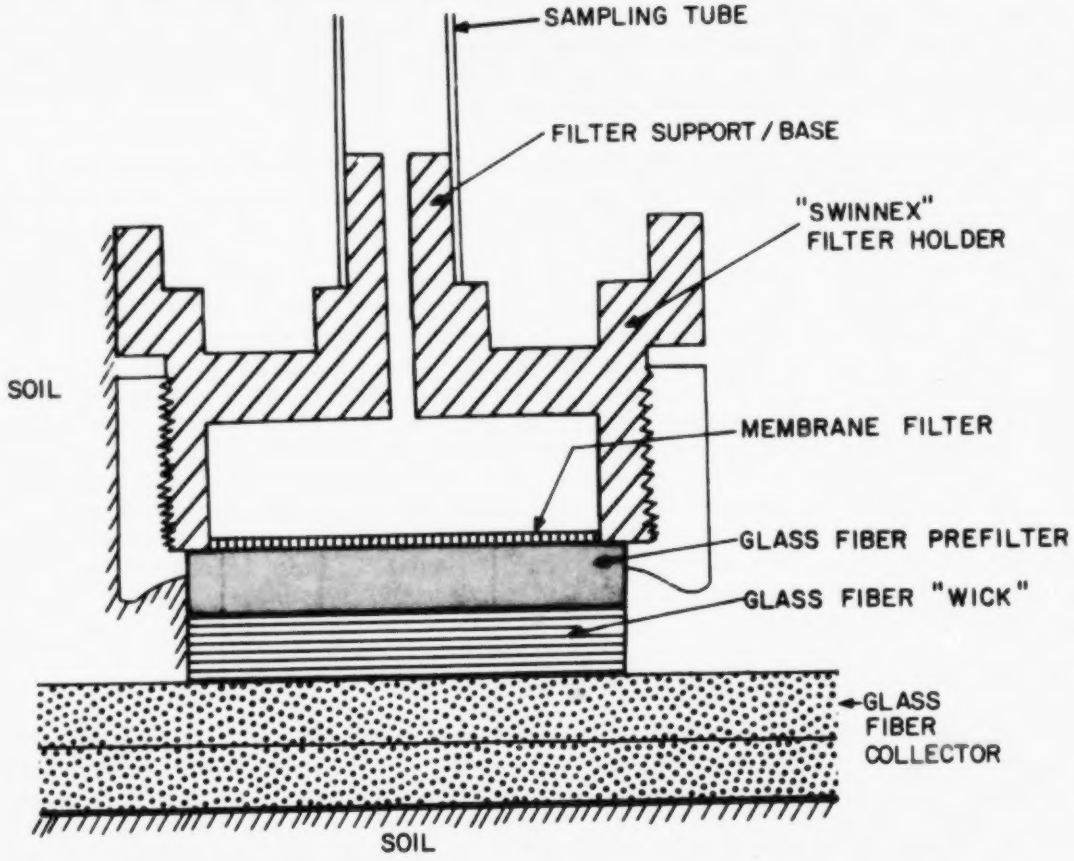

SAMPLING TUBE
FILTER SUPPORT / BASE
"SWINNEX" FILTER HOLDER
SOIL
MEMBRANE FILTER
GLASS FIBER PREFILTER
GLASS FIBER "WICK"
GLASS FIBER COLLECTOR
SOIL

Figure 38. Membrane filter sampler (after Stevenson, 1978).

The sampler is placed in a hole dug to a selected depth. Sheets of glass fiber "collectors" are placed in the bottom of the hole. Next, two or three smaller glass fiber "wick" discs that fit within the filter holder are placed in the hole. Subsequently, the filter holder is placed in the hole with the glass fiber prefilter in the holder contacting the "wick" discs. The hole is then backfilled.

In operation, soil water is drawn into the collector system by capillarity. Subsequently, water flows in the collector sheets toward the glass fiber wicks as a result of the suction applied to filter holder assembly. The glass fiber prefilter minimizes clogging of the membrane filter by fine material in the soil solution.

During field tests with the sampler, it was observed that sampling rates were a function of soil-water content. That is, sampling rates decreased with decreasing soil-water content. The "wick and collector" system provided contact with a relatively large area of the soil and a favorable sampling rate was maintained even when the "collector" became blocked with fine soil. The basic sampling unit could be used to depths of 4 meters.

Water Sampling from Saturated Regions of the Vadose Zone

Saturated regions of the vadose zone which could be monitored for water quality mainly comprise perched groundwater occurring at the interface of regions of varying texture. Perched groundwater bodies generally vary greatly in areal extent and may be either ephemeral or exist for long periods of time. For convenience, perched groundwater will be arbitrarily subdivided into the following two categories: (1) shallow perched groundwater within 30 feet of the surface, and (2) deeper perched groundwater at depths greater than 30 feet of the surface.

Inasmuch as perched groundwater is in essence saturated groundwater, many of the techniques apply to monitoring in perched groundwater (e.g., Todd et al., 1976). A guiding principle for selecting monitoring methods for both the saturated zone and perched groundwater was aptly stated by Vanhof, Weyer, and Whitaker (1979):

> ... for an efficient, long-term operation of an operational monitoring network, the devices to be used must be simple, rugged, and foolproof. The inside diameter of the installed structure should allow for the insertion of down-hole tools for development, pumping, groundwater sampling, hydraulic head measurement, and completion zone maintenance. The installed equipment should be simple enough to be used by trained but not educationally skilled personnel.

Shallow Perched Groundwater--

Six alternative methods for obtaining groundwater samples from shallow perched groundwater will be discussed: (1) tile lines, (2) collection pans and manifolds, (3) wells, (4) piezometers, (5) multilevel samplers, and (6) groundwater profile samplers. Following a discussion of each of these methods, alternative materials for well casing are reviewed and sample extraction methods are outlined.

Sampling Tile Drain Outflow--

If a tile drainage system has been installed to control the elevation of a perched groundwater table, samples could be collected from the tile outfall. In some cases, it may be desirable to install commercially available composite or discrete sampling devices.

Willardson, Meek, and Huber (1973) discussed a "flow-path groundwater sampler" which enables collection of water in different flow paths around a tile.

Collection Pans and Manifolds--

Collection pans and manifolds could be installed in regions where temporary saturation lenses develop during the percolation of water for overlying surface sources. Generally, pans are installed to permit sampling pore water through a vertical profile, usually several meters in thickness. In construction, pan collectors are similar to the vacuum extractors except that a ceramic filter candle is not included. Thus, the media in the trough must be saturated.

Commonly, the collector pans are connected to a central chamber or trench containing sample bottles and accessories. McMichael and McKee (1966) used a large-diameter culvert as the central chamber in their studies on wastewater reclamation at Whittier Narrows. Parizek and Lane (1970) used a trench to collect samples from pans in a forest soil during effluent irrigation studies in Pennsylvania.

The collectors developed by McMichael and McKee (1966) employed pan samplers for the collection of water during periods of saturation in a profile to about 10 feet. These samplers were conical in shape, 24 inches in diameter and 9 inches deep. The samplers were installed to various depths at a radial distance of about 10 feet from the central well. Individual sampling pans were filled with gravel to prohibit clogging. Each sampler was connected to the central culvert via tubing. Excavations used for installation of the samplers were carefully backfilled. During sampling, percolate intercepted by the pans flowed by gravity to the central well into collection flasks.

The trench lysimeter described by Parizek and Lane (1970) used pan lysimeters. The largest such lysimeter consisted of a 47-inch wide by 13-foot long trench excavated to about 20 feet. The sides of the trench were lined with wood and braced. Sampling pans were installed at 12-inch intervals to a depth of about 20 feet. The pans, constructed from galvanized metal, were 12 inches by 18 inches with copper tubing soldered into a raised end to permit sample drainage. Each unit was driven into the soil a short distance from the side walls of the lysimeter. While applying effluent during land treatment studies, samples of percolate intercepted by the pans were collected in flasks within the lysimeter.

A simple manifold collector was installed within a sanitary landfill by Wilson and Small (1973) for the collection of leachate (Figure 39). The collector consisted of 20 feet of 2-inch diameter PVC functioning as a horizontal collector manifold connected by a section of PVC pipe to a 4-inch diameter length of vertical PVC pipe. The unit was installed within the base of a new landfill cell prior to introducing solid waste into the area. The manifold was drilled with a large number of openings to permit entrance of leachate. The manifold was joined to the upright well about 4 inches above the base of the pipe. The base of the well was sealed so that the bottom 4 inches functioned as a storage reservoir. The manifold drain line was sloped toward the well to facilitate drainage. Leachate collecting in the reservoir was hand bailed.

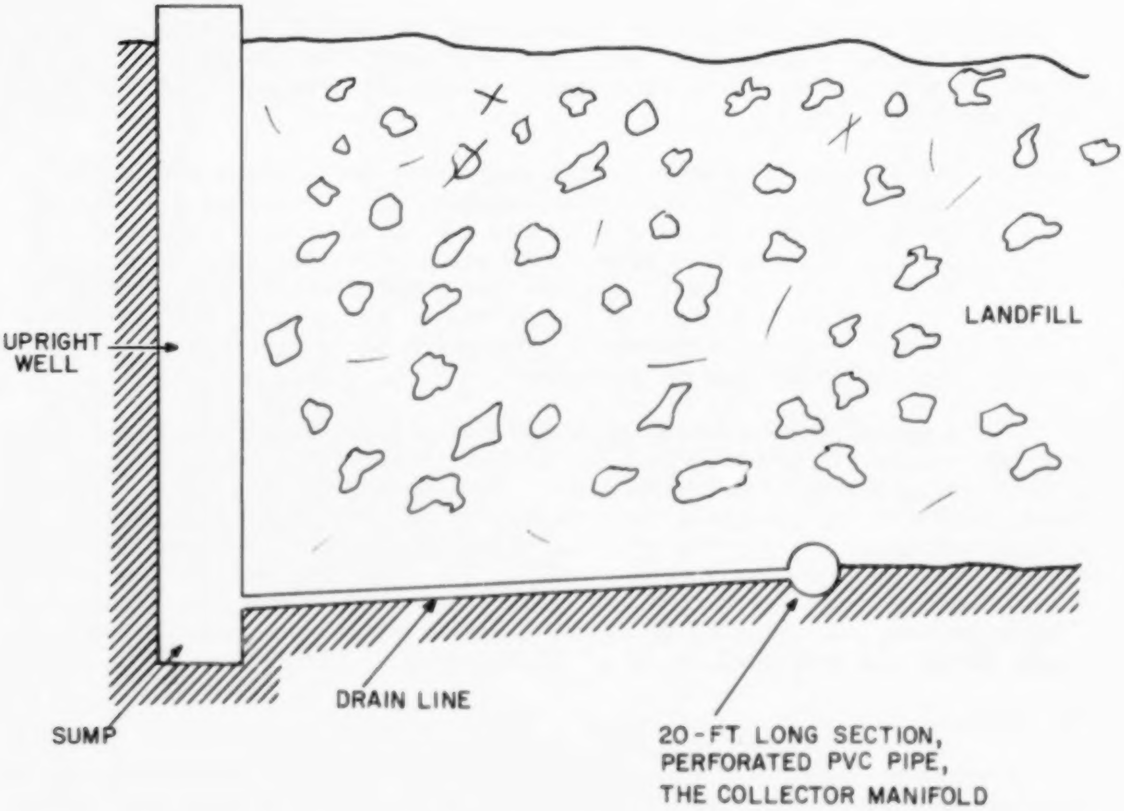

UPRIGHT WELL
LANDFILL
SUMP
DRAIN LINE
20-FT LONG SECTION, PERFORATED PVC PIPE, THE COLLECTOR MANIFOLD

Figure 39. Leachate collector installed at base of sanitary landfill.

A basic problem with collectors such as those discussed above is that samples are available only when the soil-water pressure is greater than atmospheric. During drainage, the soil-water system may shift rapidly to the unsaturated state, prohibiting sample collection.

Wells--

Wells installed to monitor water level fluctuations could also be used for sample collection. Construction techniques for installing wells were discussed earlier. However, special care must be exercised to thoroughly develop wells used for water quality monitoring. Development will ensure removal of drillers' mud and foreign water used in the drilling process. Development should be continued until sediment-free water is pumped from the well (Geraghty and Miller, 1977). Techniques for well development are thoroughly discussed in "Manual of Water Well Construction Practices" (U.S. EPA, no date). Geraghty and Miller also point out that monitoring wells must be correctly oriented in relation to the flow direction of polluted groundwater emanating from a source. A suggested practice is to locate at least one well upgradient of the source, one or more wells on site, and several wells downgradient of the source.

Observation wells should be unperforated throughout the vadose zone overlying the perched groundwater zone. Perforations below the water table should extend throughout the entire saturated thickness for reasons stated by Geraghty and Miller (1977):

> The vertical placement of the monitoring well screen within the aquifer is ... critical. For example, wells screened near the top of the zone of saturation may not detect contamination migrating in natural gradient flow paths along the base of an aquifer. In contrast, lighter-than-water constituents, such as hydrocarbons, may move along the top of the zone of saturation, thereby evading detection by monitoring wells screened only in the bottom portion of an aquifer.

It is generally recognized that wells with fully penetrating perforated sections provide information on the "bulk" or "average" concentration of pollutants in groundwater as a pumped unit (Pickens and Grisak, 1979). In many cases, a profile of pollutant distribution is also of importance because, as pointed out above, a layering of constituents may occur. The sampling units discussed below (i.e., piezometers, multilevel samplers, and profile samplers) are useful for determining pollutant stratification. Depth-wise samples can also be obtained in wells using packer pumps, such as the Casee Sampler. Packer pumps are discussed in detail by Fenn et al. (1977).

Piezometers--

Piezometer wells used for characterizing the vertical head distribution in a perched groundwater mass could also be used for water sampling. Water samples from a piezometer array with individual well points terminating at different depths will indicate the vertical displacement of indigenous groundwater with the recharge source. Similarly, if a number of arrays are distributed in a horizontal transect, samples from individual units terminating at the same elevation will manifest quality changes during the lateral spread of a pollution plume.

Methods for installing piezometers were discussed in an earlier section. Units in a given array may either be drilled separately or clustered in a common borehole. For the individual-type array, each piezometer should be thoroughly developed and cleaned in accordance with sound well construction practices. For the clustered units, development of the borehole could be difficult, but at any rate the gravel envelope around each well point should effectively prevent the entrance of fines into the well.

The location of individual arrays should be based on a thorough knowledge of the local hydrogeology, such as the slope of the water table, hydraulic conductivity, and flow velocity. Generally, wells are installed at three locations: upgradient of the source, on-site, and downgradient of the source.

Multilevel Samplers--

Pickens et al. (1978) presented details of a multilevel sampling device, shown on Figure 40. The sampler is particularly suited for sampling in cohesionless deposits with shallow groundwater in which flow occurs predominantly in the horizontal direction. The unit is installed in a single borehole and permits groundwater sampling at several levels. Hydraulic head measurements may also be obtained using a mercury manometer.

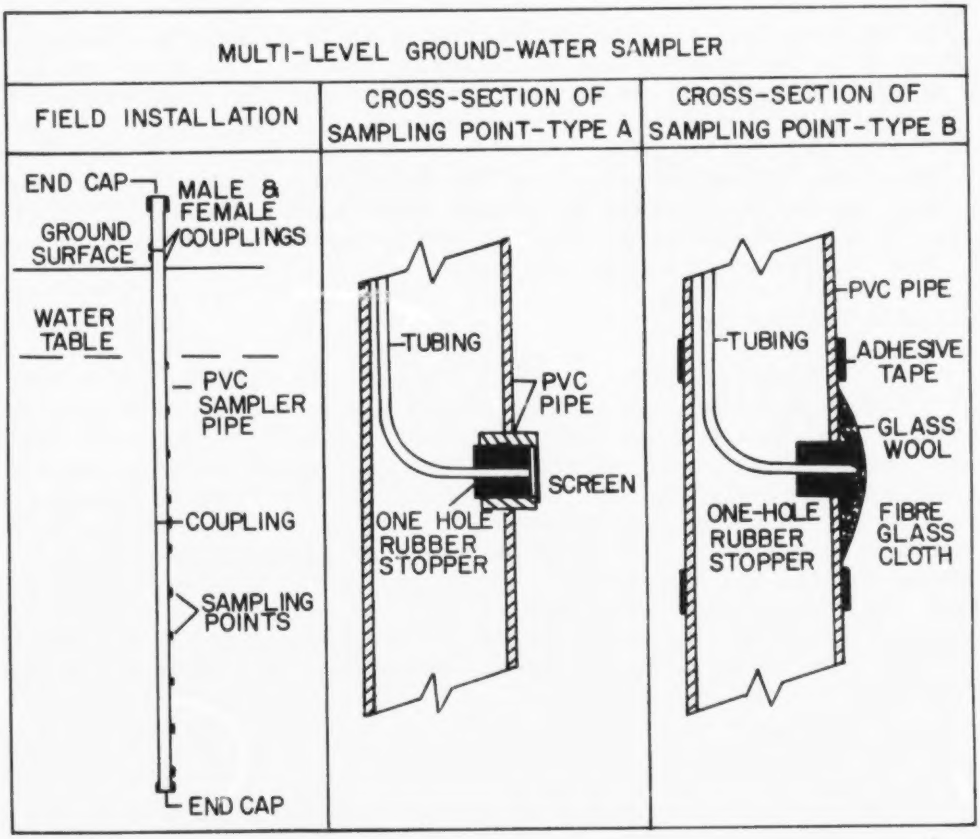

MULTI-LEVEL GROUND-WATER SAMPLER
FIELD INSTALLATION
CROSS-SECTION OF SAMPLING POINT-TYPE A
CROSS-SECTION OF SAMPLING POINT-TYPE B
END CAP
MALE & FEMALE COUPLINGS
GROUND SURFACE
WATER TABLE
PVC SAMPLER PIPE
COUPLING
SAMPLING POINTS
END CAP
TUBING
PVC PIPE
SCREEN
ONE HOLE RUBBER STOPPER
PVC PIPE
TUBING
ADHESIVE TAPE
GLASS WOOL
ONE-HOLE RUBBER STOPPER
FIBRE GLASS CLOTH

Figure 40. Multilevel sampler (after Pickens et al., 1978).

Basically, the sampler consists of the following components: (1) PVC pipe, (2) ports or openings at desired incremental depths, (3) screened covering on the openings, and (4) polypropylene tubing sealed into the openings within the pipe. The polypropylene tubing extends to the surface. A unit may be designed in the field by locating the position of openings using stratigraphic information from drilling.

The pipe can be installed by common drilling techniques such as hollow-stem auger and cable-tool. Using the hollow-stem auger, the casing is installed within the auger when the desired depth is reached and the auger is withdrawn. For cable-tool construction, it is necessary to drive in a steel casing to the desired depth, install the PVC pipe within the casing, and then pull out the casing. For both methods, it is presumed that the deposits surrounding the borehole will collapse around the casing, ostensibly preventing side leakage. Vanhof, Weyer, and Whitaker (1979) questioned this assumption. However, in their paper, Pickens et al. indicated that sealing should not be a problem with cohesionless deposits. For installation in cohesive deposits, they recommend that bentonite or grout be installed between openings to prevent cross-contamination.

During sampling, a vacuum is applied to the polypropylene tubing and water flows from the formation to a collection bottle. The water can be pumped into an air-free cell for measurement of pH and Eh. Quick delivery from the aquifer to the cell probably minimizes degassing of the water.

One of the stated advantages of the multilevel sampler is that concentration profiles can be obtained at a lower cost than by installing piezometer nests. In a subsequent paper, Pickens and Grisak (1979) present comparative cost data to substantiate this contention.

Groundwater Profile Sampler--

Hansen and Harris (1974) designed a depth-wise sampler, which they designated a "groundwater profile sampler." The sampler, shown on Figure 41, consists of a 1.25-inch diameter well point, of optional length, with isolated chambers containing fiberglass probes. The individual chambers are filled with sand and separated by caulking compound. Small-diameter tubing provides surface access to the probes. The positioning of probes is optional, depending on aquifer materials and desired sampling frequency. In operation, a vacuum is applied to the tubing, pulling groundwater into individual sampling flasks. Hansen and Harris recommended that all samples should be extracted simultaneously and at the same rate to minimize variations in aquifer thickness sampled by the individual probes. Groundwater at depths to 30 feet may be sampled with this unit.

Well Casing Material--

Two important considerations when designing wells and piezometers as quality monitoring facilities are casing material and casing diameter. A number of materials are available for well casing including plain steel, galvanized steel, PVC, fiberglass, and Teflon. Selection of material for wells at a specific site will depend on the constituents being monitored. When monitoring for salinity where only the major constituents are being evaluated, plain steel or galvanized steel could be used. When monitoring for trace

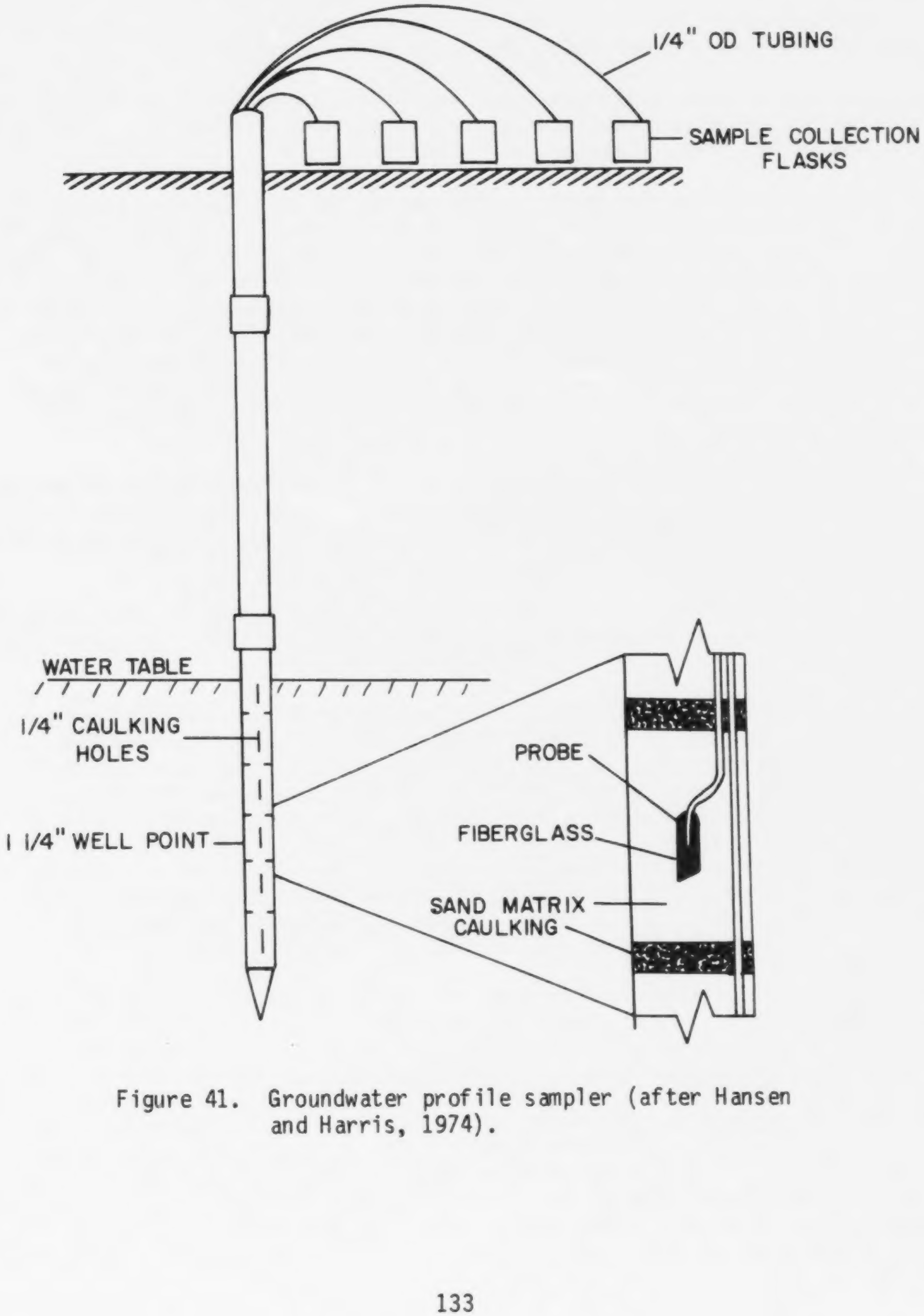

1/4" OD TUBING
SAMPLE COLLECTION FLASKS
WATER TABLE
1/4" CAULKING HOLES
1 1/4" WELL POINT
PROBE
FIBERGLASS
SAND MATRIX
CAULKING

Figure 41. Groundwater profile sampler (after Hansen and Harris, 1974).

metals, contamination from steel casing could preclude the quantification of specific constituents, such as iron and zinc. Consequently, casing constructed from PVC, fiberglass, or Teflon is advisable. Because of cost, PVC is commonly used. This material has the disadvantage that it cannot be driven into place, such as when using the cable-tool method.

A controversy exists over the type of casing material to install when monitoring for organic pollutants. For example, Dunlap et al. (1977) oppose the use of PVC pipe, stating:

> In some earlier work ... PVC casing was utilized for casing of sampling wells. This material is relatively inexpensive and easy to use, but it is less desirable as a casing material than the Teflon tubing-galvanized pipe combination for two reasons. First, organic constituents of groundwater may be adsorbed on the PVC casing. Second, there is evidence that PVC casing may contribute low levels of organic contaminants to the samples, such as phthalic acid esters used as plasticizers in PVC manufacture and solvent from cements used to join lengths of PVC tubing.

For their sampling wells, Dunlap et al. used a combination of galvanized pipe and Teflon tubing. The galvanized pipe extended from ground surface to about 1 foot above the water table. Teflon tubing was used in the portion of the well below the water table.

In contrast to the findings of Dunlap et al. regarding interaction of PVC pipe with organic constituents, Geraghty and Miller (1977) indicated the following:

> PVC pipe was used for all well casing. It is light and easy to handle and is more inert toward dissolved organic substances than steel casing. The iron oxide coating that develops on steel casing has an unpredictable and changeable adsorption capacity. However, when the adsorption sites on PVC are saturated, water remains in equilibrium with it. Leakage of organic compounds from PVC is negligible. As a control, samples of pipe and a cemented joint were submitted to the laboratory where they were soaked in water and the water was analyzed. No contaminants were detected.

No general recommendations are available on selecting well casing material when monitoring for microorganisms and virus. Conceivably, metal constituents in steel wells could inhibit the growth of microorganisms as dissolution of the casing occurs. For this reason, PVC wells are probably more suitable for sample collection when microbial levels are being evaluated. It is also probably advisable to disinfect the well following well completion. Chlorine in granular or liquid form is commonly used as a disinfectant (U.S. EPA, no date). The "Manual of Water Well Construction Practices" gives details on disinfection practices. Care must be exercised in using chlorine when excessive levels of organic constituents are present because of the possible formation of chlorinated hydrocarbons. In addition, the well should be thoroughly pumped after a suitable contact time using sterilized pumping equipment.

Selection of well casing diameter should be based on the water extraction method, quantity of sample required, budget limitations, and other uses of the well. If large quantities of sample are required and a submersible pump is to be installed, the minimum diameter of the casing should be 6 inches, and 8 inches is preferred (Schmidt, personal communication, 1979). If other extraction methods such as hand bailers, gas-lift pumps, or peristaltic pumps are suitable, smaller-diameter wells could be used. However, many wells are also used for monitoring water level changes and permanent fixtures such as gas pressure lines may occupy part of the interior space. For practical reasons, therefore, the casing diameter should probably not be less than 4 inches.

Sample Extraction Methods--

A number of techniques are available for extracting groundwater from wells and piezometers. Fenn et al. (1977) reviewed several of these methods in detail. A brief discussion is presented here of the following methods: (1) hand bailers, (2) air-lift and gas-lift pumps, (3) suction-lift pumps, (4) piston pumps, (5) centrifugal pumps, and (6) submersible pumps.

Hand bailers--In their simplest form, hand bailers consist of a pipe closed at one end and open at the other. A cable is attached to the open end to permit raising or lowering the bailer. Weights are added to the bottom of the bailer to overcome buoyancy. To collect a sample, the bailer is lowered into a well down to and below the water table. The filled container is then raised to the surface.

A more elaborate type of bailer is shown on Figure 42. This unit consists of a tube open at both ends but with a one-way check valve at the base. Basic units, such as the one shown in the figure, use a glass marble as part of the valve assembly. In operation, the unit is lowered by cable down to and below the water table. Water flows up and into the sampler body. When the unit is raised, the glass marble falls into a conical seat effectively plugging the end of the bailer. Bailers may be constructed from available materials such as steel pipe, PVC tubing, or Teflon tubing. Selection of material depends on the constituents being examined. The unit shown on the figure was constructed of Teflon to sample for highly volatile organic pollutants (Dunlap et al., 1977). The bailer is sterilized by autoclaving when sampling for microorganisms.

Another type of hand bailer is the "Kemmerer" sampler. These units are commonly used for sampling in lakes but are also suitable for groundwater sampling. The units basically consist of a cylinder with spring-loaded rubber stoppers at each end. The unit is lowered to the desired depth below the water table and a weighted "messenger" is sent down a cable to spring shut the stoppers. The sample is thus trapped within the container.

Bailers are not particularly effective for flushing casing prior to sampling unless the volume of water in the casing is minimal.

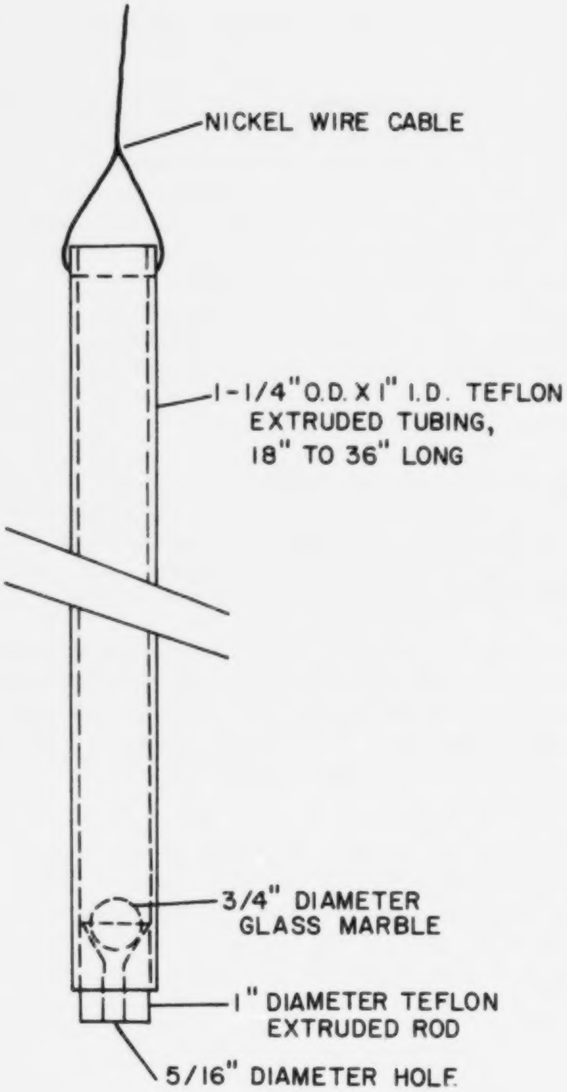
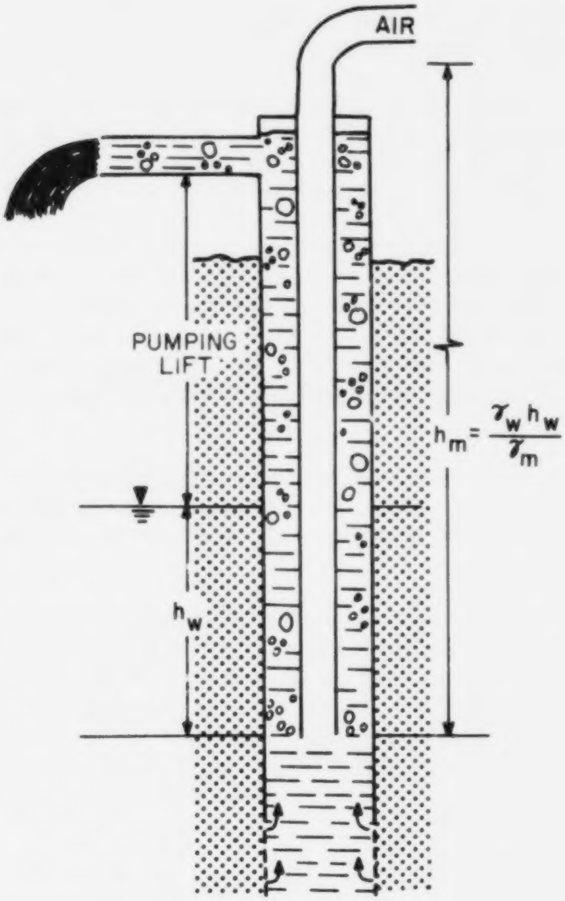

NICKEL WIRE CABLE
1-1/4" O.D. X 1" I.D. TEFLON EXTRUDED TUBING, 18" TO 36" LONG
3/4" DIAMETER GLASS MARBLE
1" DIAMETER TEFLON EXTRUDED ROD
5/16" DIAMETER HOLE

Figure 42. Hand bailer made of Teflon (after Dunlap et al., 1977).

Air-lift and gas-lift pumps--A basic air-lift pump is illustrated on Figure 43. As shown, compressed air introduced into a central pipe bubbles up through the column of water in the well. The specific gravity of the column of water is consequently reduced and an air-water mixture is discharged from an eductor pipe. In the case shown, the well casing is the eductor pipe. The principles of air-lift pumping are discussed in "Ground Water and Wells" (Johnson, 1966), and by Anderson (1977), and Trescott and Pinder (1970). As pointed out in "Ground Water and Wells," the air-lift method is particularly effective when the submergence ratio of the air line is about 60 percent. The submergence ratio is the percentage of the total length of air line that is below water when pumping. For example, if the air line is 180-feet long and


AIR
PUMPING LIFT
$h_m = \frac{\gamma_w h_w}{\gamma_m}$
h_w

h_m = MAXIMUM HEIGHT TO WHICH THE AIR-WATER MIXTURE WILL RISE

h_w = SUBMERGED LENGTH OF THE AIR LINE

γ_m = SPECIFIC WEIGHT OF THE AIR-WATER MIXTURE

γ_w = SPECIFIC WEIGHT OF WATER

▼ = POTENTIOMETRIC SURFACE

Figure 43. Schematic representation of simple air-lift pump showing principle of operation (after Trescott and Pinder, 1970).

the water level when pumping is at 74 feet, the submergence rate is (180-74)/180 or 59 percent. Apparently, favorable pumping rates are still possible with as little as 30-percent submergence.

Trescott and Pinder (1970) developed a simple air-lift pump for sampling from small-diameter piezometers. An automobile engine was used to provide air pressure for the pump. Components of the pump included: a ball check valve, an air line, a pressure gage, and a header assembly on the well. The check valve which screws into the spark plug socket of an engine cylinder operates such that clean air from outside the engine is sucked into the cylinder during the inlet stroke. The compressed air is then forced into the air line during the compression stroke. The check valve maintains pressure in the air line. The header assembly which is fastened to the top of the well casing contains an inlet opening, through which the air line is inserted, and a discharge pipe. The discharge rate for air-lift pumps increases with increasing pipe diameter.

If sufficient air is available and the submergence rate is favorable, enough water can be extracted by the air-lift method for sampling purposes. The system could also be used for the initial development of the well and for flushing prior to sampling.

Gronowski (1979) described a new commercial pump operating on the same principle as the air-lift pump except that the pumping energy is obtained from 14-ounce propane cylinders. According to Gronowski, "The unit consists of dual-conductor plastic tubing with a valve assembly on top for attachment to the propane cylinder. A shaped copper tubing serves as the intake assembly. Propane injected down one side of the tubing forces water to rise up the other side to ground surface. Several samples of adequate volume can be collected with one 14-ounce propane cylinder."

Bianchi, Johnson, and Haskell (1962) designed an alternative air-lift pump which in effect is an automated hand bailer. As shown on Figure 44, the unit consists of a cylinder with a flat one-way valve at the base and a two-hole rubber stopper at the upper end. One nylon tube is inserted through one of the holes and extended to the lower end of the cylinder. A second nylon line is inserted into the second opening. In operation, the cylinder is lowered below the water table and water flows upward through the valve. Air pressure is then applied to the shorter line, the one-way valve shuts, and water is forced up the longer line to the surface. A tire pump could be used to apply pressure. Samples have been lifted with this unit from depths greater than 100 feet.

The simple air-lift pumps introduce air into the sample. Consequently, extracted samples are unsuitable for determining dissolved gasses and many unstable constituents in groundwater. The effect of propane on such constituents, when using the gas-lift pump, should be examined before the method is employed in a specific monitoring program.

Suction-lift pumps--The basic principle of suction-lift pumps is that, when a vacuum is created inside a tube or pipe extending below the water table, air pressure on the water outside the tube forces water to rise inside

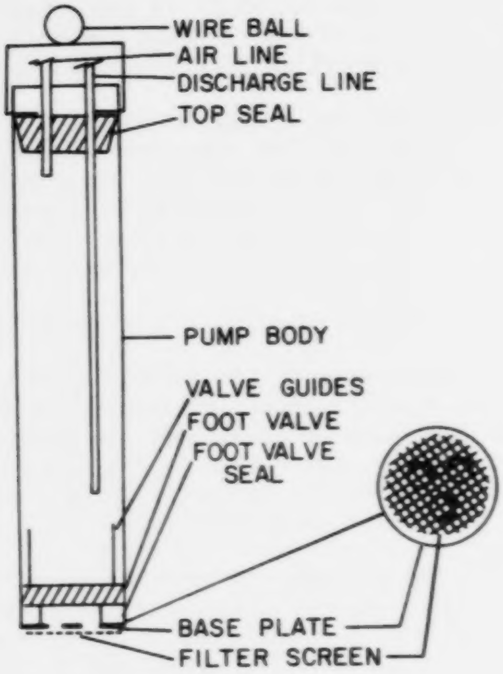


Figure 44. Positive action air-lift pump (after Bianchi, Johnson, and Haskell, 1962).

the tube (Anderson, 1977). The practical suction lift with such pumps is about 25 feet. Allison (1971) discussed a simple suction-lift pump for extracting groundwater from auger holes. The method could also be used for cased wells. Basically, the unit consists of plastic tubing of sufficient length to extend below the water table. The upper end of the tube terminates in a collector flask. A second line from the flask is connected to a hand-held vacuum pump. By applying vacuum to the system, water is forced into the flask.

A variation of the suction-lift pump is the so-called peristaltic pump, or squeeze pump. The pump consists of a rotor equipped with rotating heads at both ends. The rotor is mounted in a closed housing. Plastic tubing is placed between the rotor heads and the housing such that when the rotor turns, the plastic tubing is squeezed by the rotating heads. The tubing is lowered into a well below the water table and primed with water. When the unit is activated, the squeezing action of the heads produces a negative pressure on the water in the tubing extending into the well. At the same time, a positive pressure is exerted on the water in the tubing leaving the housing.

An advantage of the peristaltic pump is that water is in contact only with the tubing. Contamination of the water sample with pump accessories is thus minimized. Dunlap et al. (1977) used peristaltic pumps for both grab sampling and continuous extraction of groundwater when sampling for organic pollutants.

A basic problem with peristaltic pumps is that discharge rates are very low. Also, foreign water is added when priming the unit. The latter problem could be overcome by using hand-bailed groundwater for priming.

Piston pumps--Piston pumps are a type of positive displacement pump often used in windmill wells. Basically, the pump consists of a cylinder containing a plunger or piston which is driven up and down via a rod. The lower end of the cylinder contains an inlet valve consisting of a one-way check valve. A one-way check valve is also mounted in the piston. The piston also contains exterior leathers which contact the walls of the cylinder.

The operation of the pump is described in "Ground Water and Wells":

> As the piston moves downward, the intake valve closes at the instant that the pressure above it exceeds the pressure below it; and the discharge valve opens when the pressure below it exceeds that above it. Water thus is trapped in the cylinder during the downstroke of the piston, then is forced upward into the discharge pipe on the next upstroke.

During installation, the cylinder is attached to a length of piston rod inside a length of PVC pipe. The PVC pipe functions as an eductor tube. Successive lengths of pipe and rod are added until the cylinder is located at the desired depth below the water table. The upper end of the pipe/rod assembly is rigidly attached to the casing. The PVC pipe could be attached to a T-section, permitting the sample to be discharged into a collector flask. When the water table is not in excess of about 10 feet, the pump may be operated by hand. For deeper water tables, a tripod and pulley arrangement may be necessary. Alternatively, a pitcher pump could be attached to the top of the well.

In addition to providing water samples, piston pumps are an effective method for developing a new well. Also, if the volume of water stored in the well is not excessive, piston pumps may be used for flushing the casing prior to sampling. A problem with piston pumps for routine sampling is that the installation time may be excessive compared to more portable units. An expensive alternative would be to permanently install a separate unit in each monitoring well. Another basic problem with piston pumps is that metals from the cylinder may contaminate pumped samples.

Centrifugal pumps--Engine-driven centrifugal pumps could be used to extract water samples from lifts of about 15 feet. Depending on well diameter, such pumps could be used to extract large quantities of samples.

Submersible pumps--Submersible pumps comprise a sealed centrifugal-type pumping plant including an electric motor which operates below the water table. Water is pumped to the surface via delivery tubing.

McMillion and Keeley (1968) described a portable submersible pump suitable for sampling in wells greater than 4.5 inches inside diameter. The components of the unit are a submersible pump, 300 feet of flexible hose, an electric winch and spool assembly, and a portable generator. The generator provides electricity to drive the pump and operate the winch. In operation, the hose is unreeled from the spool assembly to position the pump at the desired depth and the pump is activated. Pumping rates range from 7 to 14 gallons per minute. The assembly is stored in a van during transit between wells.

Although the submersible pump assembly of McMillion and Keeley is expensive, it is convenient to install and operate within a well, the time for sample collection is minimal, and the unit permits rapid flushing of stagnant water in the casing prior to sampling. According to Fenn et al. (1977), if there are no leaks in the system, the unit could be used to sample for dissolved gasses.

Well Flushing Prior to Sampling--

Sound monitoring practice requires that a monitor well or piezometer be thoroughly flushed prior to sample collection. The rationale is that groundwater within a passive monitor well tends to be affected by exposure to the atmosphere and by mixing within the casing. [Passive monitoring is defined as periodically sampling wells in the path of groundwater flow for changes in concentration of the constituents of interest (Hart Associates, 1979)]. Consequently, in order to obtain a truly representative sample of water from the perched region, the water in the casing should be completely displaced.

Fenn et al. (1977) thoroughly reviewed procedures for flushing. The principal recommendations are cited below:

> To safeguard against collecting nonrepresentative stagnant water in a sample, the following guidelines and techniques should be adhered to during sample withdrawal: As a general rule, all monitoring wells should be pumped or bailed prior to withdrawing a sample. Evacuation of a minimum of one volume of water in the well casing and preferably three to five volumes is recommended for a representative sample. In a high-yielding groundwater formation and where there is no stagnant water in the well above the screened section, evacuation prior to sample withdrawal is not as critical. However, in all cases where the monitoring data are to be used for enforcement actions, evacuation is recommended.
>
> For wells that can be pumped or bailed to dryness with the sampling equipment being used, the well should be evacuated and allowed to recover prior to sample withdrawal. If the recovery rate is fairly rapid and time allows, evacuation of more than one volume of water is preferred.
>
> For high-yielding monitoring wells which cannot be evacuated to dryness, bailing without prepumping the well is not recommended; there is no absolute safeguard against contaminating the sample with stagnant water. The following procedures should be used:

(a) The inlet line of the sampling pump should be placed just below the surface of the well water and three to five volumes of water pumped at a rate equal to the well's recovery rate. This provides reasonable assurance that all stagnant water has been evacuated and that the sample will be representative of the groundwater body at that time. The sample can then be collected directly from the pump discharge line.

(b) The inlet line of the sampling pump (or the submersible pump itself) should be placed near the bottom of the screen section, pumped approximately one well volume of water at the well's recovery rate, and the sample collected directly from the discharge line.

A nonrepresentative sample can also result from excessive prepumping of the monitoring well. Stratification of the leachate concentrations in the groundwater formation may occur, and excessive pumping can dilute or increase the contaminant concentrations from what is representative of the sampling point of interest.

Techniques for Sampling Perched Groundwater in the Lower Vadose Zone--

In deep vadose zones, such as those of western alluvial basins, it is not unusual to encounter one or more regions of perched groundwater in the lower vadose zone. Water samples from these deeper saturated regions reflect the quality of water which has moved below the soil zone enroute to the water table. In many cases, large quantities of water can be extracted from these deeper perched groundwater zones. Furthermore, the samples may reflect the integrated quality of water draining from an extensive portion of the overlying vadose zone. This will be particularly true when the overlying source is diffuse, such as irrigation return flow. Because of the integrated nature of the water in these perched regions, samples may be obtained using one or more wells at a fraction of the cost for installing batteries of suction cups.

Wilson and Schmidt (1979) presented data from a number of case studies on monitoring perched groundwater. The sampling techniques reviewed below are discussed in detail in their paper. Two methods are possible for sampling perched groundwater: (1) cascading water, and (2) special wells. Generally, perched groundwater cascades into wells through poorly welded joints, cracks, or perforations exposed as the water table recedes. Figure 45 illustrates a cross section of a conceptualized well with cascading water. Samples of cascading water could be obtained from abandoned wells in the vicinity of the source being monitored. Alternatively, cascading water could be obtained from operating wells whenever pumps are removed for servicing.

Samples of cascading water can be obtained by lowering a bucket to the desired depth. In sampling for chemical constituents, the sample container and sample bottles should be rinsed several times during sample collection. To sample for microorganisms, a sterilized container should be used. Inasmuch as cascading water is exposed to the atmosphere, some problems may be

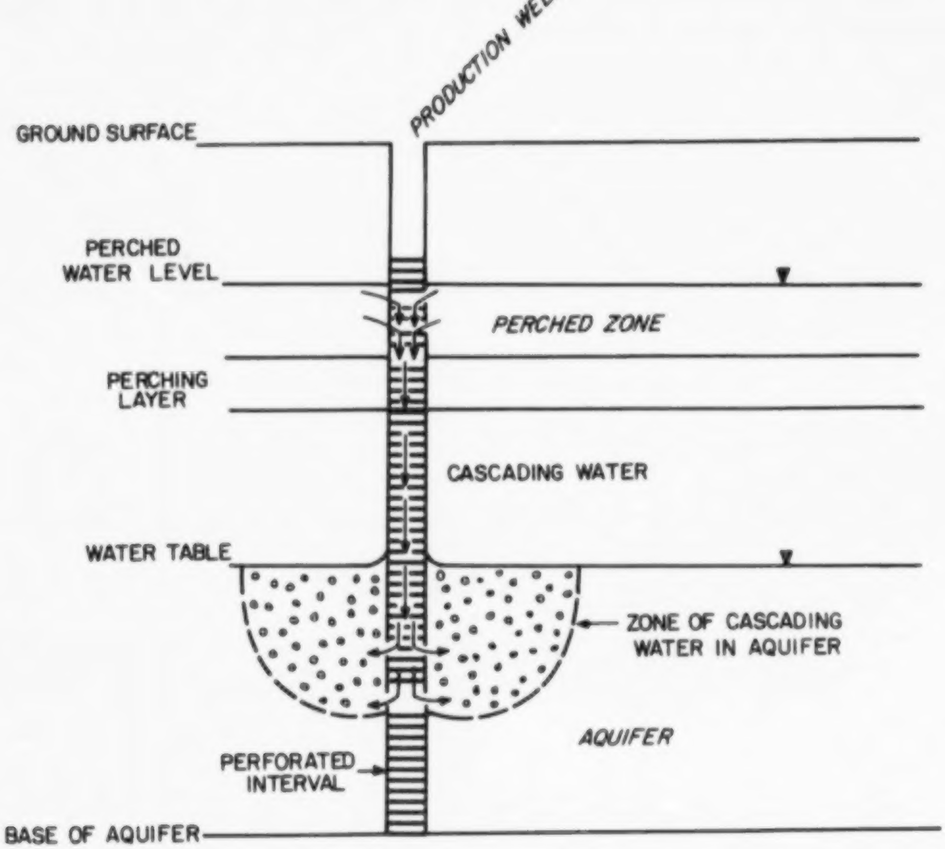

PRODUCTION WELL
GROUND SURFACE
PERCHED WATER LEVEL
PERCHED ZONE
PERCHING LAYER
CASCADING WATER
WATER TABLE
ZONE OF CASCADING WATER IN AQUIFER
AQUIFER
PERFORATED INTERVAL
BASE OF AQUIFER

Figure 45. Conceptualized cross section of a well showing cascading water from perched zone (after Wilson and Schmidt, 1979).

experienced in determining pH, dissolved oxygen, and oxidation-reduction potential.

If wells with cascading water are not present in the vicinity of a source, it will be necessary to construct special wells (Figure 46). A major problem will be to locate wells at the proper depth to tap perched water. Alternative methods for locating existing or potential perching regions were discussed earlier in this report, including drillers' observations, grain size of drill cuttings, and borehole geophysical methods.

Materials for casing will depend on the constituents of interest in the monitoring program. Thus, if metals and possibly organic pollutants are of concern, it may be advisable to use PVC casing. Inasmuch as PVC casing cannot be driven, the rotary technique will be required for installation when the region of interest is deep. If the region is not too deep, however, it may be possible to use the cable-tool method as follows: steel casing is driven into the borehole as drilling progresses and, when the desired depth is reached, PVC casing is lowered into the steel casing. The steel casing is then pulled out, leaving the PVC casing in place. The PVC casing should be preperforated throughout the region of interest.

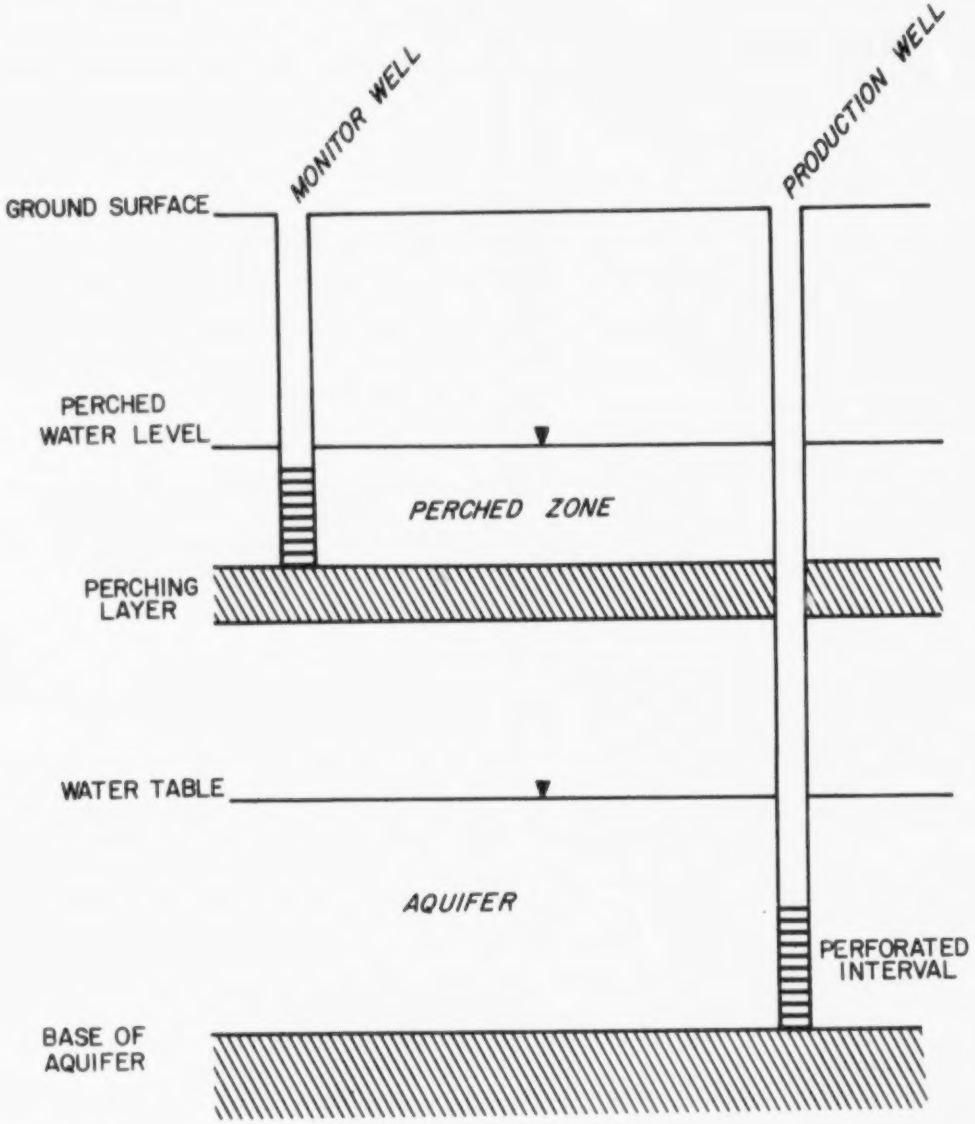

MONITOR WELL
PRODUCTION WELL
GROUND SURFACE
PERCHED WATER LEVEL
PERCHED ZONE
PERCHING LAYER
WATER TABLE
AQUIFER
PERFORATED INTERVAL
BASE OF AQUIFER

Figure 46. Conceptualized monitor well used to sample from a perched zone (after Wilson and Schmidt, 1979).

The ideal casing size for monitoring wells is probably about 8 inches (Schmidt, personal communication, 1979). This size permits easy installation and removal of a portable submersible pump for sampling. In addition, a water level recorder could also be installed within 8-inch casing.

When several perching layers occur, a cluster of sampling wells could be installed within a common borehole. Methods for clustering wells are identical to those discussed elsewhere for multiple piezometer installations.

Several limitations in sampling perched groundwater should be noted. Perched zones do not occur everywhere and may not be present at a specific area of interest. The detection and location of perched groundwater may be expensive, involving the construction of test wells or requiring the use of geophysical methods as discussed above. Even when found, some perched groundwater bodies may be ephemeral. Alternatively, certain perching regions may be more responsive to recharge from a given source of pollution than other regions. Thus, the location of successful monitoring wells may in part be a matter of chance.

A problem in sampling cascading wells is that water may enter the casing from a number of distinct perching regions. Consequently, the water collected below the uppermost region could be a mixture of water from various depths. However, such a blend could provide useful information.

REFERENCES

Alemi, M.H., D.R. Nielsen, and J.W. Biggar, Determining the Hydraulic Conductivity of Soil Cores by Centrifugation, Journal Soil Science Soc. America, 40, 42-218, 1976.

Alesii, B.A., Cyanide Mobility in Soils, unpublished M.S. Thesis, The University of Arizona, Tucson, Arizona, 1976.

Alexander, M., Introduction to Soil Microbiology, John Wiley and Sons, Inc., New York, 1961.

Allison, L.E., A Simple Device for Sampling Ground Waters in Auger Holes, Soil Sci. Soc. Amer. Proc., 35, 844-845, 1971.

Anderson, K.E., editor, Water Well Handbook, Missouri Water Well Drillers Assocation, Rolla, Missouri, 1977.

Apgar, M.A., and D. Langmuir, Ground-Water Pollution Potential of a Landfill Above the Water Table, Ground Water, 9(6), 76-93, 1971.

Ayers, R.S., and D.W. Westcot, Water Quality for Agriculture, Irrigation and Drainage Paper No. 29, Food and Agriculture Organization of the United Nations, 1976.

Ayers, R.S., and R.L. Branson, editors, Nitrates in the Upper Santa Ana River Basin in Relation to Groundwater Pollution, California Agric. Exp. Station, Bulletin 861, 1973.

Bailey, G.W., and J.L. White, Factors Influencing the Adsorption, Desorption, and Movement of Pesticides in Soil, in Residue Reviews, Vol. 32, Springer-Verlag, New York, 1970.

Bear, J., Dynamics of Fluids in Porous Media, American Elsevier, New York, 1972.

Bear, J., D. Zaslavsky, and S. Irmay, Physical Principles of Water Percolation and Seepage, United National Educational, Scientific and Cultural Organization, 465 pp, 1968.

Bertrand, A.R., Rate of Water Intake in the Field, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 197-208, American Society of Agronomy, Madison, Wisconsin, 1965.

Bianchi, W.C., C.E. Johnson, and E.E. Haskell, A Positive Action Pump for Sampling Small Bore Wells, *Soil Sci. Soc. Amer. Proc.*, 26(1), 86-87, 1962.

Bianchi, W., Measuring Soil Moisture Tension Changes, *Agricultural Engineering*, 43(7), 398, 399, 404, 1967.

Biggar, J.W., and D.R. Nielsen, Spatial Variability of the Leaching Characteristics of a Field Soil, *Water Resources Research*, 12(1), 78-84, 1976.

Black, C.A., editor, *Methods of Soil Analyses*, (in two parts), Agronomy No. 9, American Society of Agronomy, Madison, Wisconsin, 1965.

Blake, G.R., Bulk Density, in *Methods of Soil Analyses*, C.A. Black, editor, Agronomy No. 9, 374-390, American Society of Agronomy, Madison, Wisconsin, 1965.

Bodman, G.B., and E.A. Coleman, Moisture and Energy Conditions During Downward Entry of Water into Soils, *Soil Sci. Soc. Amer. Proc.*, 8, 116-122, 1944.

Bohn, H.L., B.L. McNeal, and G.A. O'Connor, *Soil Chemistry*, Wiley Interscience, New York, New York, 1979.

Bordner, R., J. Winter, and P. Scarpino, *Microbiological Methods for Monitoring the Environment*, EPA-600/8-78-017, U.S. Environmental Laboratory, Environmental Monitoring and Support Laboratory, 1978.

Borg, I.Y., R. Stone, H.B. Levy, and L.D. Ramspott, *Information Pertinent to the Migration of Radionuclides in Ground Water at the Nevada Test Site, Part I: Review and Analysis of Existing Information*, Lawrence Livermore Laboratory, Livermore, California, National Technical Information Service, 1976.

Bouwer, H., *Groundwater Hydrology*, McGraw-Hill Book Co., New York, 480 pp, 1978.

Bouwer, H., and R.D. Jackson, Determining Soil Properties, in *Drainage for Agriculture*, J. van Schilfgaarde, editor, Agronomy No. 17, 611-675, American Society of Agronomy, Madison, Wisconsin, 1974.

Bower, C.A., and L.V. Wilcox, Soluble Salts, in *Methods of Soil Analyses*, C.A. Black, editor, Agronomy No. 9, 933-951, American Society of Agronomy, Madison, Wisconsin, 1965.

Brakensiek, D.L., Empirical and Simplified Models of the Infiltration Process, in *Infiltration Research Planning Workshop, Part 1, State of the Art Reports*, Sciences and Education Administration, The U.S. Dept. of Agric., ARM-NC-4, 1979.

Brakensiek, D.L., H.B. Osborn, and W.J. Rawls, *Field Manual for Research in Agricultural Hydrology*, Agricultural Handbook No. 224, Sciences and Education Administration, The U.S. Dept. of Agric., 1979.

Broadbent, F.E., Organics, in Proceedings of the Joint Conference on Recycling Municipal Sludges and Effluents on Land, sponsored by The Environmental Protection Agency, The U.S. Dept. of Agric., and The Nat. Assoc. of State Univ. and Land-Grant Colleges, Champaign, Illinois, July 9-13, 1973.

Cartwright, K., and F.B. Sherman, Electrical Earth Resistivity Surveying in Landfill Investigations, Proceedings 10th Annual Engineering and Soils Engineering Symposium, Moscow, Idaho, 77-92, 1972.

Cartwright, K., and M.R. McComas, Geophysical Surveys in the Vicinity of Sanitary Landfills in Northeastern Illinois, Ground Water, 6(5), 23-30, 1968.

Cary, J.W., Soil Water Flowmeters with Thermocouple Outputs, Soil Sci. Soc. Amer. Proc., 37, 176-181, 1973.

Chang, A.C., and A.L. Page, Hydraulic and Water Quality Considerations of Artificial Groundwater Recharge, in Proceedings of the Twelfth Biennial Conference on Ground Water, Sacramento, September 20-21, 1979, California Water Resources Center, The University of California, Davis, Report No. 45, pp 88-99, 1979.

Chapman, H.D., Cation Exchange Capacity, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 891-900, American Society of Agronomy, Madison, Wisconsin, 1965.

Childs, E.C., An Introduction to the Physical Basis of Soil Water Phenomena, Wiley Interscience, New York, New York, 1969.

Chow, T.L., A Porous Cup Soil-Water Sampler with Volume Control, Soil Science, 124, 173-176, 1977.

Coelho, M.A., Spatial Variability of Water Related Soil Physical Properties, unpublished M.S. Thesis, The University of Arizona, Tucson, Arizona, 1974.

Cooley, R.L., J.F. Harsh, and D.C. Lewis, Principles of Ground-Water Hydrology, Hydrologic Engineering Methods for Water Resources Development, Vol. 10, The Hydrologic Engineering Center, U.S. Army Corps of Engineers, Davis, California, 1972.

Corey, J.C., S.F. Peterson, and M.A. Wakat, Measurement of Attenuation of ^{137}Cs and ^{241}Am Gamma Rays for Soil Density and Water Content Determinations, Soil Sci. Soc. Amer. Proc., 35, 215-219, 1971.

Davis, J.B., Microbial Decomposition of Hydrocarbons, Industrial and Engineering Chemistry, 48(9), 1444-1448, 1956.

Davis, S.N., and R.J.M. deWiest, Hydrogeology, John Wiley and Sons, Inc., New York, 1966.

Day, P.R., G.H. Bolt, and D.M. Anderson, Nature of Soil Water, in *Irrigation of Agricultural Lands*, R.M. Hagan, H.R. Haise, and T.W. Edminster, editors, *Agronomy No. 9*, 193-208, American Society of Agronomy, Madison, Wisconsin, 1967.

Dirksen, C., Water Flux Measurements in Unsaturated Soils, in *Flow-Its Measurement and Control in Science and Industry*, Vol. 1, 479-486, Inst. Soc. Amer., Pittsburgh, Pennsylvania, 1974a.

Dirksen, C., Field Test of Soil Water Flux Meters, *Trans. Amer. Soc. Agric. Eng.*, 17(6), 1038-1042, 1974b.

Donnan, W.W., Field Methods, in *Drainage of Agricultural Lands*, J.N. Luthin, editor, American Society of Agronomy, Madison, Wisconsin, 1957.

Duke, H.R., and H.R. Haise, Vacuum Extractors to Assess Deep Percolation Losses and Chemical Constituents of Soil Water, *Soil Sci. Soc. Amer. Proc.*, 37, 963-964, 1973.

Dunlap, W.J., and J.F. McNabb, *Subsurface Biological Activity in Relation to Ground Water Pollution*, EPA-660/2-73-014, U.S. Enviornmental Protection Agency, Corvallis, Oregon, 1973.

Dunlap, W.J., J.F. McNabb, M.R. Scalf, and R.L. Cosby, *Sampling for Organic Chemicals and Microorganisms in the Subsurface*, EPA-600/2-77-176, U.S. Environmental Protection Agency, 1977.

Ellis, B.G., The Soil as a Chemical Filter, in *Recycling Treated Municipal Waste Water and Sludge Through Forest and Cropland*, W.E. Sopper and L.T. Kardos, editors, The Pennsylvania State University Press, 1973.

Enfield, C.G., J.J.C. Hsieh, and A.W. Warrick, Evaluation of Water Flux Above a Deep Water Table Using Thermocouple Phsychrometers, *Soil Sci. Soc. Amer. Proc.*, 37(6), 968-970, 1973.

England, C.B., Comments of "A Technique Using Porous Cups for Water Sampling at Any Depth in the Unsaturated Zone" by Warren W. Wood, *Water Resources Research*, 10(6), 1049, 1974.

Estes, J.E., D.S. Simonett, L.R. Tinney, C.E. Ezra, B. Bowman, and M. Roberts, *Remote Sensing Detection of Perched Water Tables*, Technical Completion Report, Contribution No. 175. California Water Resources Center, The University of California, Davis, California, 1978.

Evans, D.D., T.W. Sammis, A.W. Warrick, *Transient Movement of Water and Solutes in Unsaturated Soil Systems*, Phase II, Project Completion Report, OWRT Project No. B040-ARIZ, Department of Hydrology and Water Resources, The University of Arizona, Tucson, Arizona, 1976.

Everett, L.G., editor, Groundwater Quality Monitoring of Western Coal Strip Mining: Identification and Priority Ranking of Potential Pollution Sources, EPA-600/7-79-024, U.S. Environmental Protection Agency, Las Vegas, Nevada, 1979.

Everett, L.G., K.D. Schmidt, R.M. Tinlin, and D.K. Todd, Monitoring Groundwater Quality: Methods and Costs, EPA-600/4-76-023, U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Las Vegas, Nevada, 1976.

Fenn, D., E. Cocozza, J. Isbister, O. Briads, B. Yare, and P. Roux, Procedures Manual for Ground Water Monitoring at Solid Waste Disposal Facilities, EPA/530/SW-611, U.S. Environmental Protection Agency, 1977.

Fenn, D.G., K.J. Hanley, and T.V. DeGeare, Use of the Water Balance Method for Predicting Leachate Generation from Solid Waste Disposal Sites, EPA/530/SW-618, U.S. Environmental Protection Agency, Office of Solid Waste, 1975.

Freeze, R.A., The Mechanism of Natural Ground-Water Recharge; 1 - One-Dimensional Vertical, Unsteady, Unsaturated Flow Above a Discharging Ground-Water Flow System, Water Resources Research, 5(1), 153-171, 1969.

Freeze, R.A., and J.A. Cherry, Groundwater, Prentice-Hall Inc., Englewood Cliffs, New Jersey, 1979.

Frissel, M.J., P. Poelstra, K. Harmsen, and G.H. Bolt, Tracing Soil-Moisture Migration with ^{36}Cl, ^{60}Co and Tritium, in Isotope and Radiation Techniques in Soil Physics and Irrigation Studies 1973, International Atomic Energy Agency, Vienna, 1974.

Fuller, W.H., Movement of N-Selected Metals, Asbestos, and Cyanide in Soil: Application to Waste Disposal Problems, EPA-600/2-77-020, U.S. Environmental Protection Agency, 1977.

Fuller, W.H., Investigation of Landfill Leachate Pollutant Attenuation by Soils, EPA-600/2-78-158, U.S. Environmental Protection Agency, Cincinnati, Ohio, 1978.

Gairon, S., and A. Hadas, Measurement of the Water Status in Soils, in Arid Zone Irrigation, B. Yaron, E. Danfois, and Y. Vaadia, editors, 215-226, Springer-Verlag, New York, 1973.

Gardner, W.H., Water Content, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 82-125, American Society of Agronomy, Madison, Wisconsin, 1965.

Geraghty & Miller, Inc., The Prevalence of Subsurface Migration of Hazardous Chemical Substances at Selected Industrial Waste Land Disposal Sites, EPA/530/SW-634, U.S. Environmental Protection Agency, 1977.

Gilbert, R.G., C.P. Gerba, R.C. Rice, H. Bouwer, C. Wallis, and J.L. Melnick, Virus and Bacteria Removal from Wastewater by Land Treatment, *Applied and Environmental Microbiology*, 32(3), 333-338, 1976.

Green, W.H., and G.A. Ampt, Studies on Soil Physics: I. Flow of Air and Water Through Soils, *Journal of Agricultural Research*, 4, 1-24, 1911.

Gronowski, R., Innovations Simplify Shallow Well Monitoring, *Water Well Journal*, 33(7), 57, 1979.

Guma'a, G.S., *Spatial Variability of In-Situ Available Water*, unpublished Ph.D. Dissertation, The University of Arizona, Tucson, Arizona, 1978.

Hall, W.A., Theoretical Aspects of Water Spreading, *Amer. Soc. Ag. Eng.*, 36(6), 394-397, 1955.

Hamon, W.R., Infiltrometer Using Simulated Rainfall for Infiltration Research, in *Infiltration Research Planning Workshop, Part 1, State of the Art Reports*, Sciences and Education Administration, The U.S. Dept. of Agric., ARM-NC-4, 1979.

Hansen, E.A., and A.R. Harris, A Groundwater Profile Sampler, *Water Resources Research*, 10(2), 375, 1974.

Hansen, E.A., and A.R. Harris, Validity of Soil-Water Samples Collected With Porous Ceramic Cups, *Soil Sci. Soc. Amer. Proc.*, 39, 528-536, 1975.

Hantush, M.A., Growth and Decay of Ground-Water Mounds in Response to Uniform Percolation, *Water Resources Research*, 3(1), 277-234, 1967.

Hart, Fred C., Associates, *Draft Environmental Impact Statement on the Proposed Guidelines for the Landfill Disposal of Solid Waste*, U.S. Environmental Protection Agency, Office of Solid Waste, 1979.

Hem, J.D., *Study and Interpretation of the Chemical Characteristics of Natural Water*, U.S. Geological Survey Water Supply Paper, 1473, 1970.

Hillel, D., *Soil and Water Physical Principles and Processes*, Academic Press, New York, 1971.

Hoffman, G.J., C. Dirksen, R.D. Ingvalson, E.V. Maas, J.D. Oster, S.L. Rawlins, J.D. Rhoades, and J. van Schilfgaarde, Minimizing Salt in Drain Water by Irrigation Management, in *Agricultural Water Management*, 1, 233-252, Elsevier Scientific Publishing Company, Amsterdam, 1978.

Holmes, J.W., S.A. Taylor, and S.J. Richards, Measurement of Soil Water, in *Irrigation of Agricultural Lands*, R.M. Hagan, H.R. Haise, and T.W. Edminster, editors, Agronomy No. 9, 275-298, American Society of Agronomy, Madison, Wisconsin, 1967.

Hounslow, A., J. Fitzpatrick, L. Cerrillo, and M. Freeland, Overburden Mineralogy as Related to Ground-Water Chemical Changes in Coal Strip Mining, EPA-600/7-78-156, U.S. Environmental Protection Agency, Ada, Oklahoma, 1978.

Huibregtse, K.R., and J.H. Moser, Handbook for Sampling and Sample Preservation of Water and Wastewater, EPA-600/4-76-049, U.S. Environmental Protection Agency, Quality Assurance Branch, Cincinnati, Ohio, 1976.

Jackson, D.R., F.S. Brinkley, and E.A. Bondietti, Extraction of Soil Water Using Cellulose-Acetate Hollow Fibers, Journal Soil Science Soc. America, 40, 327-329, 1976.

Jackson, R.D., R.J. Reginato, and C.H.M. van Bavel, Comparison of Measured and Calculated Hydraulic Conductivities of Unsaturated Soils, Water Resources Research, 1, 375-379, 1965.

Jakubick, A.T., Migration of Plutonium in Natural Soils, in Transuranium Nuclides in the Environment, International Atomic Energy Agency, Vienna, 1976.

Jensen, M.E., Consumptive Use of Water and Irrigation Water Requirements, American Society of Civil Engineers, 1973.

Johnson, E.E., Inc. Ground Water and Wells, E.E. Johnson, Inc., St. Paul, Minnesota, 1966.

Jurinak, J.J., and J. Santillan-Medrano, The Chemistry and Transport of Lead and Cadmium in Soils, Research Report No. 18, Agric. Exp. Station, Utah State University, 1974.

Keeney, D.R., and R.E. Wildung, Chemical Properties of Soils, in Soils for Management of Organic Wastes and Waste Water, Soil Sci. Soc. Amer., American Soc. of Agron., Crop Science Soc. of America, Madison, Wisconsin, 1977.

Keys, W.S., The Application of Radiation Logs to Ground-Water Hydrology, in International Atomic Energy Agency Symposium on Use of Isotopes in Hydrology, 477-488, 1967.

Keys, W.S., and L.M. MacCary, Application of Borehole Geophysics to Water Resources Investigations, in Technical Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 2, 126 pp, 1971.

Kirkham, D., Studies of Hillside Seepage in the Iowa Drift Area, Soil Sci. Soc. Amer. Proc., 12, 73-80, 1947.

Kirkham, D., and W.L. Powers, Advanced Soil Physics, Wiley Interscience, New York, New York, 1972.

Klute, A., Laboratory Measurement of Hydraulic Conductivity of Unsaturated Soil, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 253-261, American Society of Agronomy, Madison, Wisconsin, 1965.

Klute, A., The Movement of Water in Unsaturated Soils, in The Progress of Hydrology, Proc. First Int. Seminar for the Hydrology Prof., The National Science Foundation Science Seminar, Dept. of Civil Eng., The University of Illinois, July 13-25, 1969.

Kokotov, Y.A., R.F. Popova, and A.P. Urbanyuk, Sorption of Long-Lived Fission Products on Soils and Clay Minerals, Radiokhimiya, 3(2), 199-206, 1961.

Korte, N.E., J. Skopp, W.H. Fuller, E.E. Niebla, and B.A. Alesii, Trace Element Movement in Soils: Influence of Soil Physical and Chemical Properties, Soil Science, 122(5), 350-359, 1976.

Kraijenhoff van deLeur, D.A., Some Effects of the Unsaturated Zone on Nonsteady Free-Surface Groundwater Flow as Studied in a Scaled Granular Model, J. Geophys. Res., 67(11), 4347-4362, 1962.

LaRue, M.E., D.R. Nielsen, and R.M. Hagan, Soil Water Flux Below a Ryegrass Root Zone, Agronomy Journal, 60, 625-629, 1968.

Leenheer, J.A., and E.W.D. Huffman, Jr., Classification of Organic Solutes in Water by Using Macroreticular Resins, Journal Research U.S. Geol. Survey, 4(6), 737-751, 1976.

LeGrand, H.E., System for Evaluating the Contamination Potential of Some Waste Sites, Journal American Water Works Association, 56(8), 959-974, 1964.

Leonard, R.A., G.W. Bailey, and R.R. Swank, Transport, Detoxification, Fate, and Effects of Pesticides in Soil and Water, in Land Application of Soils Material, Soil Conservation Society of America, 1976.

Levin, M.J., and D.R. Jackson, A Comparison of In-Situ Extractors for Sampling Soil Water, Journal Soil Science Soc. America, 41, 535-536, 1977.

Lissey, A., The Use of Reducers to Increase the Sensitivity of Piezometers, Journal of Hydrology, 5(2), 197-205, 1967.

Lohman, S.W., Ground-Water Hydraulics, U.S. Geological Survey Professional Paper No. 708, 1972.

Lund, L.J., and P.F. Pratt, Variability of Nitrate Leaching Within Defined Management Units, in Proceedings of the National Conference on Irrigation Return Flow Quality Management, Colorado State University, 1977.

Lusczynski, N.J., Filter-Press Method of Extracting Water Samples for Chloride Analysis, U.S. Geological Survey Water Supply Paper, 1544-A, 1961.

Luthin, J.M., editor, Drainage of Agricultural Lands, American Society of Agronomy, Madison, Wisconsin, 1957.

Luthin, J.M., and P.R. Day, Lateral Flow Above a Sloping Water Table, Soil Sci. Soc. Amer. Proc., 19, 406-410, 1955.

Mann, J.F., Wastewaters in the Vadose Zone of Arid Regions: Hydrologic Interactions, Ground Water, 14(6), 367-372, 1976.

Mather, J.R., and P.A. Rodriguez, The Use of the Water Budget in Evaluating Leaching Through Solid Waste Landfills, Final Report on OWRT Project No. A-040-DEL, Water Resources Center, The University of Deleware, Newark, Deleware, 1978.

Matlock, W.G., G.C.A. Morin, and J.E. Posedly, Well Cuttings Analysis in Ground-Water Resources Evaluation, Ground Water, 14(5), 272-277, 1976.

Matlock, W.G., and P.R. Davis, Groundwater in the Santa Cruz Valley, Technical Bulletin 194, Agric. Exp. Station, The University of Arizona, Tucson, Arizona, 1972.

McMichael, F.C., and J.E. McKee, Wastewater Reclamation at Whittier Narrows, Publication No. 33, State of California, Water Quality, 1966.

McMillion, L.G., and J.W. Keeley, Sampling Equipment for Groundwater Investigations, Ground Water, 6(2), 9-11, 1968.

McNeal, B.L., Soil Salts and Their Effect on Water Movement, in Drainage for Agriculture, J. van Schilfgaarde, editor, Agronomy No. 17, 409-433, American Society of Agronomy, Madison, Wisconsin, 1974.

McWhorter, D.B., and J.A. Brookman, Pit Recharge Influenced by Subsurface Spreading, Ground Water, 10(5), 6-11, 1972.

Meinzer, Oscar E., Ground Water, in Hydrology, Oscar E. Meinzer, editor, 385-477, Dover Publications, Inc., New York, 1942.

Merrill, S.D., and S.C. Rawlins, Field Measurement of Soil Water Potential with Thermocouple Psychrometers, Soil Science, 113(2), 102-109, 1972.

Meyer, W.R., Use of a Neutron Moisture Probe to Determine the Storage Coefficient of an Unconfined Aquifer, U.S. Geological Survey, Professional Paper 450-E, 174-176, 1963.

Miller, R.H., The Soil as a Biological Filter, in Recycling Treated Municipal Waste Water and Sludge Through Forest and Cropland, W.E. Sopper and L.T. Kardos, editors, The Pennsylvania State University Press, 1973.

Millington, R.J., and J.P. Quirk, Permeability of Porous Media, Nature, 183, 387-388, 1959.

Mooij, H., and F.A. Rovers, Recommended Groundwater and Soil Sampling Procedures, Environmental Protection Service, Report EPS-4-EC, Canada, 1976.

Mortland, M.M., and W.D. Kemper, Specific Surface, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 532-543, American Society of Agronomy, Madison, Wisconsin, 1965.

Murrmann, R.P., and F.R. Koutz, Role of Soil Chemical Processes in Reclamation of Wastewater Applied to Land, in Wastewater Management by Disposal on the Land, U.S. Army Corps of Engineers, Cold Regions Research and Eng. Lab. Specialty Report 171, 1972.

Myhre, D.L., J.O. Sanford, and W.F. Jones, Apparatus and Techniques for Installing Access Tubes in Soil Profiles to Measure Soil Water, Soil Science, 108(4), 296-299, 1969.

Nelson, J.L., Soil Column Studies with Radiostrontium: I. Effects of Temperature of Species of Accompanying Ion, H.W. 62035, Hanford Atomic Products Operation, Richland, Washington, 1959.

Nielsen, D.R., and J.W. Biggar, Analyzing Soil Water and Solute Movement Under Field Conditions, in Soil-Moisture and Irrigation Studies II, International Atomic Energy Agency, 1973.

Nielsen, D.R., W. Biggar, and K.T. Erh, Spatial Variability of Field Measured Soil-Water Properties, Hilgardia, 43(7), 215-260, 1973.

Nommik, H., Ammonium Fixation and Other Reactions Involving a Nonenzymatic Immobilization of Mineral Nitrogen in Soil, in Soil Nitrogen, W.V. Barthalemew and F.E. Clark, editors, Agronomy No. 10, 1965.

Norris, S.E., The Use of Gamma Logs in Determining the Character of Unconsolidated Sediments and Well Construction Features, Ground Water, 10(6), 14-21, 1972.

Ongerth, H.J., D.P. Spath, J. Crook, and A.F. Greenberg, Public Health, Aspects of Organics in Water, Journal American Water Works Association, 65, 495, 1973.

Oster, J.D., and L.S. Willardson, Reliability of Salinity Sensors for the Management of Soil Salinity, Agronomy Journal, 63, 695-698, 1971.

Paetzold, R.F., Measurement of Soil Physical Properties, in Infiltration Research Planning Workshop, Part 1, State of the Art Reports, Sciences and Education Administration, The U.S. Dept. of Agric., ARM-NC-4, 1979.

Parizek, R.R., L.J. Kardos, W.E. Sopper, E.A. Myers, D.E. Dairs, M.A. Farrell, and J.B. Nesbitt, Waste Water Renovation and Conservation, Pennsylvania State Studies, No. 23, The Pennsylvania State University Press, 1967.

Parizek, R.R., and B.E. Lane, Soil-Water Sampling Using Pan and Deep Pressure-Vacuum Lysimeters, Journal of Hydrology, 11 1-21, 1970.

Pavoni, J.L., D.J. Hagerty, and R.E. Lee, Water Resources Bulletin, 8(6), 1091-1107, 1972.

Phene, C.J., G.J. Hoffman, and S.L. Rawlins, Measuring Soil Matric Potential In-Situ by Sensing Heat Dissipation Within a Porous Body: I Theory and Sensor Construction, Soil Sci. Soc. Amer. Proc., 35, 27-33, 1971.

Phene, C.J., S.L. Rawlins, and G.J. Hoffman, Measuring Soil Matric Potential In-Situ by Sensing Heat Dissipation Within a Porous Body: II Experimental Results, Soil Sci. Soc. Amer. Proc.,35, 225-229, 1971.

Philip, J.R., Theory of Infiltration, in Advances in Hydroscience, 5, 216-269, 1969.

Pickens, J.F., and G.E. Grisak, Reply to the Preceding Discussion of Vanhof et al. at "A Multilevel Device for Groundwater Sampling and Piezometric Monitoring", Ground Water, 17(4), 393-397, 1979.

Pickens, J.R., J.A. Cherry, G.E. Grisak, W.F. Merritt, and G.A. Risto, A Multilevel Device for Ground-Water Sampling and Piezometric Monitoring, Ground Water, 16(5), 322-327, 1978.

Pratt, P.F., W.W. Jones, and V.E. Hunsaker, Nitrate in Deep Soil Profiles in Relation to Fertilizer Rates and Leaching Volume, Jour. Envir. Quality, 1(1), 97-102, 1972.

Pratt, P.F., J.E. Warneke, and P.A. Nash, Sampling the Unsaturated Zone in Irrigated Field Plots, Journal Soil Science Soc. America, 40, 277-279, 1976.

Pratt, P.F., A.C. Chang, J.P. Martin, A.L. Page, and C.F. Kleine, Removal of Biological and Chemical Contaminants by Soil Systems in Association with Ground Water Recharge by Spreading or Injection of Treated Municipal Waste Water, in Health Aspects of Wastewater Recharge, A State-of-the-Art Review, Water Information Center, Huntington, New York, 1978.

Quirk, J.P., and R.K. Schofield, The Effect of Electrolyte Concentration on Soil Permeability, J. Soil Science, 6, 1963-1978, 1955.

Ralston, D.R., Influences of Water Well Design on Neutron Logging, unpublished M.S. Thesis, The University of Arizona, Tucson, Arizona, 1967.

Rawlins, S.L., and F.N. Dalton, Psychrometric Measurement of Soil Water Potential Without Precise Temperature Control, Soil Sci. Soc. Amer. Proc., 31, 297-301, 1967.

Reeve, R.C., Hydraulic Head, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 180-196, American Society of Agronomy, Madison, Wisconsin, 1965.

Reginato, R.J., and C.H.M. van Bavel, Soil Water Measurement with Gamma Attenuation, Soil Sci. Soc. Amer. Proc., 28(6), 721-724, 1964.

Reginato, R.J., Count Rate Instability in Gamma Ray Transmission Equipment, Soil Sci. Soc. Amer. Proc., 38, 156-157, 1974.

Reginato, R.J., and R.D. Jackson, Field Measurement of Soil-Water Content by Gamma-Ray Transmission Compensated for Temperature Fluctuations, Soil Sci. Soc. Amer. Proc., 35, 529-533, 1971.

Reitemeier, R.F., Effect of Moisture Content on the Dissolved and Exchangeable Ions of Soils and Arid Regions, Soil Science, 61(3), 195-214, 1946.

Rhoades, J.D., and L. Bernstein, Chemical, Physical and Biological Characteristics of Irrigation and Soil Water, in Water and Water Pollution Handbook, Vol. 1, L.L. Ciaccio, editor, 141-222, Marcel Dekker, Inc., New York, 1971.

Rhoades, J.D., and J. van Schilfgaarde, An Electrical Conductivity Probe for Determining Soil Salinity, Soil Sci. Soc. Amer. Proc., 40, 647-651, 1976.

Rhoades, J.D., and A.D. Halvorson, Electrical Conductivity Methods for Detecting and Delineating Saline Seeps and Measuring Salinity in Northern Great Plains Soils, ARS W-42, The U.S. Dept. of Agric., 1977.

Rhoades, J.D., M.T. Kaddah, A.D. Halvorson, and J.R. Prather, Establishing Soil Electrical Conductivity-Salinity Calibrations Using Four-Electrode Cells Containing Undisturbed Soil Cores, Soil Science, 123, 137-141, 1977.

Rhoades, J.D., Monitoring Soil Salinity: A Review of Methods, in Establishment of Water Quality Monitoring Programs, American Water Resources Association, 1979a.

Rhoades, J.D., Salinity Management and Monitoring, in Proceedings of the Twelfth Biennial Conference on Ground Water, Sacramento, September 20-21, 1979, California Water Resources Center, The University of California, Davis, 1979b.

Rhoades, J.D., Inexpensive Four-Electrode Probe for Monitoring Soil Salinity, Journal Soil Science Soc. America, (in press), 1979c.

Rhodes, D.W., Adsorption of Plutonium by Soil, Soil Science, 84, 465-472, 1957a.

Rhodes, D.W., The Effect of pH on the Uptake of Radioactive Isotopes from Solution by a Soil, Soil Sci. Soc. Amer. Proc., 21, 389-392, 1957b.

Rible, J.M., P.A. Nash, P.F. Pratt, and L.H. Lund, Sampling the Unsaturated Zone of Irrigated Lands for Reliable Estimates of Nitrate Concentrations, Journal Soil Science Soc. America, 40, 566-570, 1976.

Richards, L.A., A Soil Salinity Sensor of Improved Design, Soil Sci. Soc. Amer. Proc., 30, 333-337, 1966.

Richards, L.A., editor, Diagnosis and Improvement of Saline and Alkali Soils, Agric. Handbook 60, The U.S. Dept. of Agric., 1954.

Richards, L.A., Physical Condition of Water in Soils, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 128-152, American Society of Agronomy, Madison, Wisconsin, 1965.

Robertson, J.M., C.R. Toussaint, and M.A. Jorque, Organic Compounds Entering Ground Water from a Landfill, EPA-660/2-74-077, U.S. Environmental Protection Agency, 1974.

Rolston, D.E., and F.E. Broadbent, Field Measurement of Denitrification, EPA-600/2-77-233, U.S. Environmental Protection Agency, 1977.

Rose, C.W., Agricultural Physics, Pergamon Press, 1966.

Runnells, D.D., Wastewaters in the Vadose Zone of Arid Regions: Geochemical Interactions, Ground Water, 14(6), 374-385, 1976.

Schmidt, K.D., Protection of Ground Water from Nonpoint Sources of Pollution, in Drinking Water Quality Enhancement Through Source Protection, R.B. Pojasek, editor, Ann Arbor Science Pub., Inc., Ann Arbor, Michigan, 1977.

Schmidt-Collerus, J.J., The Disposal and Environmental Effects of Carbonaceous Solid Wastes from Commercial Oil Shale Operations, University of Denver Research Institute, Denver, Colorado, 1974.

Schwille, F., Petroleum Contamination of the Subsoil - A Hydrological Problem, in The Joint Problems of Oil and Water Industries, P. Hepple, editor, Elsevier, Amsterdam, 1967.

Sciences and Education Administration, Infiltration Research Planning Workshop, Part 1, State of the Art Reports, The U.S. Dept. of Agric., ARM-NC-4, 1979.

Scott, V.H., and J.C. Scalmanini, Water Wells and Pumps: Their Design, Construction, Operation and Maintenance, Bulletin 1889, Division of Agricultural Sciences, The University of California, 1978.

Shaffer, K.A., D.D. Fritton, and D.E. Baker, Drainage Water Sampling in a Wet, Dual-Pore Soil System, Jour. Envir. Quality, 8(2), 241-246, 1979.

Shuval, H.E., and N. Gruener, Health Considerations in Renovating Wastewater for Domestic Use, Environmental Science and Tech., 7, 600, 1973.

Sidle, R.C., Infiltration Measurements and Soil Hydraulics Characteristics, in Infiltration Research Planning Workshop, Part 1, State of the Art Reports, Sciences and Education Administration, The U.S. Dept. of Agric., ARM-NC-4, 1979.

Silka, L.R., and T.L. Swearingen, A Manual for Evaluating Contamination Potential of Surface Impoundments, EPA-570/9-78-003, Ground Water Protection Branch, Office of Drinking Water, U.S. Environmental Protection Agency, 1978.

Simonson, R.W., What Soils Are, in Soil, The Yearbook of Agriculture, The U.S. Dept. of Agric., 1957.

Slawson, G.C., editor, Groundwater Quality Monitoring of Western Oil Shale Development: Identification and Priority Ranking of Potential Pollution Sources, EPA-600/7-79-023, U.S. Environmental Protection Agency, Las Vegas, Nevada, 1979.

Soil Moisture Equipment Corporation, Operating Instructions for the Model 1900 Soil Water Sampler, Santa Barbara, California, 1978.

Soiltest, Inc., Soil Testing Equipment, General Catalog, 1976.

Sopper, W.E., and L.T. Kardos, editors, Recycling Treated Municipal Waste Water and Sludges Through Forest and Cropland, The Pennsylvania State University Press, 1973.

Stallman, R.W., Flow in the Zone of Aeration, in Advances in Hydroscience, Vol. 4, Ven Te Chow, editor, Academic Press, New York, 1967.

Steele, R.G.D., and J.H. Torrie, Principles and Procedures of Statistics, McGraw-Hill Book Co., New York, 1960.

Stevenson, C.D., Simple Apparatus for Monitoring Land Disposal Systems by Sampling Percolating Soil Waters, Environmental Science and Engineering, 12(3), 329-331, 1978.

Stollar, R.L., and P. Roux, Earth Resistivity Surveys - A Method for Determining Ground-Water Contamination, Ground Water, 13(2), 145-150, 1975.

Taylor, G.S., and J.N. Luthin, The Use of Electronic Computers to Solve Subsurface Drainage Problems, Water Resources Research, 5(1), 144-152, 1969.

Thomas, G.W., and R.E. Phillips, Consequences of Water Movement in Macropores, Jour. Envir. Quality, 8(2), 149-152, 1979.

Thomas, R.E., The Soil as a Physical Filter, in Recycling Treated Municipal Wastewater and Sludge Through Forest and Cropland, W.E. Sopper and L.T. Kardos, editors, The Pennsylvania State University Press, 1973.

Thornthwaite, C.W., and J.R. Mather, Instructions and Tables for Computing Potential Evapotranspiration and Water Balance, Drexel Institute of Technology Laboratory of Climatology, Publications in Climatology, Volume X, No. 3, Centerton, New Jersey, 1957.

Todd, D.K., R.M. Tinlin, K.D. Schmidt, and L.G. Everett, Monitoring Groundwater Quality: Monitoring Methodology, EPA-600/4-76-026, U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Las Vegas, Nevada, 1976.

Topp, G.C., and W. Zebchuck, The Determination of Soil-Water Desorption Curves for Soil Cores, Canadian Journal of Soil Sci., 59, 19-26, 1979.

Trescott, P.C., and G.F. Pinder, Air Pump for Small-Diameter Piezometers, Ground Water, 8(3), 10-15, 1970.

U.S. Bureau of Reclamation, The U.S. Dept. of the Interior, Ground Water Manual, A Water Resources Technical Publication, 1977.

U.S. Environmental Protection Agency, Hazardous Waste Guidelines and Regulations, Federal Register, 43(243), Part IV, 58946-59027, 1978.

U.S. Environmental Protection Agency, Draft Application Forms for EPA Programs, Federal Register, 44(116), Part III, 34346-34416, 1979a.

U.S. Environmental Protection Agency, Landfill Disposal of Solid Waste, Proposed Guidelines, Federal Register, 44(59), Part II, 18138-18148, 1979b.

U.S. Environmental Protection Agency, Manual of Water Well Construction Practices, EPA-570/9-75-001, Office of Water Supply, no date.

U.S. Environmental Protection Agency, U.S. Army Corps of Engineers, and The U.S. Dept. of Agric., Process Design Manual for Land Treatment of Municipal Wastewater, EPA-625/1-77-008 (COE EM/110-1-501), 1977.

U.S. Salinity Laboratory Staff, Minimizing Salt in Return Flow Through Irrigation Management, EPA-600/2-77-134, U.S. Environmental Protection Agency, Ada, Oklahoma, 1977.

van Bavel, C.H.M., Neutron Scattering Measurement of Soil Moisture: Development and Current Status, Proc. Int. Symp. Humidity and Moisture, Washington, D.C., 171-184, 1963.

Vanhof, J.A., K.U. Weyer, and S.H. Whitaker, Discussion of "A Multilevel Device for Ground-Water Sampling and Piezometric Monitoring by J.F. Pickens, J.A. Cherry, G.E. Grisak, W.F. Merritt, and B.A. Rizto", Ground Water, 17(4), 391-393, 1979.

van Schilfgaarde, J., Theory of Flow to Drains, in Advances in Hydroscience, 6, 43-106, 1970.

Vomicil, J.A., Porosity, in Methods of Soil Analyses, C.A. Black, editor, Agronomy No. 9, 299-314, American Society of Agronomy, Madison, Wisconsin, 1965.

Wagner, G.H., Use of Ceramic Cups to Sample Soil Water Within the Profile, Soil Science, 94, 379-386, 1962.

Walker, W.H., Where Have All the Toxic Chemicals Gone?, Ground Water, 11(2), 11-20, 1973.

Warner, D.L., Preliminary Field Studies Using Earth Resistivity Measurements for Delineating Zones of Contaminated Ground Water, Ground Water, 7(1), 9-16, 1969.

Warrick, A.W., and A. Amoozegar-Fard, Area Prediction of Water and Solute Flux in the Unsaturated Zone, Final Report on Grant No. R-804-751-010, U.S. Environmental Protection Agency, (in press), 1980.

Watson, K.K., A Recording Field Tensiometer with Rapid Response Characteristics, Journal of Hydrology, 5, 33-39, 1967.

Weeks, E.P., Field Determination of Vertical Permeability to Air in the Unsaturated Zone, U.S. Geological Survey Professional Paper No. 1051, 1978.

Wheeler, M.L., Moisture and Solute Transport in Porous Media, in Actinides in the Environment, A.M. Friedman, editor, ACS/Symposium Series 35, American Chemical Society, 1976.

Willardson, L.S., B.D. Meek, and M.J. Huber, A Flow Path Ground Water Sampler, Soil Sci. Soc. Amer. Proc., 36, 965-966, 1973.

Wilson, L.G., Observations on Water Content Changes in Stratified Sediments During Pit Recharge, Ground Water, 9(3), 29-40, 1971.

Wilson, L.G., and J.N. Luthin, Effect of Air Flow Ahead of the Wetting Front on Infiltration, Soil Science, 96, 136-143, 1963.

Wilson, L.G., and K.J. DeCook, Field Observations on Changes in the Subsurface Water Regime During Influent Seepage in the Santa Cruz River, Water Resources Research, 4(6), 1219-1234, 1968.

Wilson, L.G., P.S. Osborne, and D.J. Percious, Dilution of an Industrial Waste Effluent with River Water in the Vadose Region During Pit Recharge, Paper No. 68-727, Presented at the Winter Meetings, American Society of Agricultural Engineers, 1968.

Wilson, L.G., and K.D. Schmidt, Monitoring Perched Ground Water in the Vadose Zone, in Establishment of Water Quality Monitoring Programs, American Water Resources Association, 1979.

Wilson, L.G., and G.G. Small, Pollution Potential of a Sanitary Landfill Near Tucson, Hydraulic Engineering and the Environment, Proceedings 21st Annual Hydraulics Division Specialty Conference, American Society of Civil Engineers, 1973.

Winograd, I.J., Radioactive Waste Storage in the Arid Zone, EOS, Transactions American Geophysical Union, 55(10), 884-894, 1974.

Wolff, R.G., Weathering Woodstock Granite Near Baltimore, Maryland, American Journal of Science, 265, 106-117, 1967.

Wood, W.W., A Technique Using Porous Cups for Water Sampling at Any Depth in the Unsaturated Zone, Water Resources Research, 9(2), 486-488, 1973.

Wood, W.W., and D.C. Signor, Geochemical Factors Affecting Artificial Recharge in the Unsaturated Zone, Trans. Amer. Soc. of Agric. Eng., 18(4), 677-683, 1975.

Zohdy, A.A.R., G.P. Eaton, and D.R. Mabey, Application of Surface Geophysics to Ground-Water Investigations, in Techniques of Water Resources Investigations of The United States Geological Survey, Chapter D1, Book 2, Collection of Environmental Data, 1974.

APPENDIX A

CONVERSION FACTORS
U.S. Customary to SI (Metric)

U.S. customary unit			SI	
Name	Abbreviation	Multiplier	Symbol	Name
acre	acre	0.405	ha	hectare
acre-foot	acre-ft	1,234	m^3	cubic metre
cubic foot	ft^3	28.32 0.0283	L m^3	litre cubic metre
cubic feet per second	ft^3/s	28.32	L/s	litres per second
degrees Fahrenheit	°F	0.555(°F-32)	°C	degrees Celsius
feet per second	ft/s	0.305	m/s	metres per second
foot (feet)	ft	0.305	m	metre(s)
gallon(s)	gal	3.785	L	litre(s)
gallons per acre per day	gal/acre•d	9.353	L/ha•d	litres per hectare per day
gallons per day	gal/d	4.381×10^{-5}	L/s	litres per second
gallons per minute	gal/min	0.0631	L/s	litres per second
horsepower	hp	0.746	kW	kilowatt
inch(es)	in.	2.54	cm	centimetre(s)
inches per hour	in./hr	2.54	cm/h	centimeters per hour
mile	mi	1.609	km	kilometre

(continued)

CONVERSION FACTORS (continued)

U.S. customary unit			SI	
Name	Abbreviation	Multiplier	Symbol	Name
miles per hour	mi/h	0.45	m/s	metres per second
million gallons	Mgal	3.785	ML	megalitres (litres x 10^6)
		3,785	m^3	cubic metres
million gallons per acre	Mgal/acre	8,353	m^3/ha	cubic metres per hectare
million gallons per day	Mgal/d	43.8	L/s	litres per second
		0.044	m^3/s	cubic metres per second
parts per million	ppm	1	mg/l	milligrams per litre
pound(s)	lb	0.454	kg	kilogram(s)
		453.6	g	gram(s)
pounds per acre per day	lb/acre•d	1.12	kg/ha•d	kilograms per hectare per day
pounds per square inch	lb/in.^{2}	0.069	kg/cm^2	kilograms per square centimetre
		0.69	N/cm^2	Newtons per square centimetre
square foot	ft^2	0.0929	m^2	square metre
square inch	in.^{2}	6.452	cm^2	square centimetre
square mile	mi^2	2.590	km^2	square kilometre
		259	ha	hectare

GLOSSARY

adjusted sodium adsorption ratio: A modified form of the sodium adsorption ratio which accounts for the interrelationships among sodium, calcium, and magnesium ions in an applied water or matrix solution, and for the tendency for calcium and magnesium ions to precipitate or dissolve in the presence of carbonate and bicarbonate.

adsorption isotherm: A graphical representation of the amount of adsorbate (solute) adsorbed by an adsorbent as a function of the equilibrium concentration of adsorbate (Bohn, NcNeal, and O'Connor, 1979).

anisotropy: A porous system is anisotropic with respect to the hydraulic conductivity (or other properties) if values of the hydraulic conductivity are not equal in all directions away from a given point.

antecedent water: Water stored in a soil prior to infiltration of applied water.

average linear velocity: The flux (specific discharge) divided by the water filled porosity; accounts for variations in the cross-sectional areas through which flow occurs in a porous system.

capillary fringe: The basal region of the vadose zone comprising sediments that are saturated, or nearly saturated, near the water table, gradually decreasing in water content with increasing elevation above the water table.

cascading water: Perched groundwater which enters a well casing via cracks or uncovered perforations, trickling or pouring down the inside of the casing.

cation exchange capacity: The total capacity of a porous system to adsorb cations from a solution.

clay micelle: An individual clay particle.

distribution coefficient: A representation of the partitioning of solids between liquid and solid phases in a porous system (Freeze and Cherry, 1979).

electrical conductivity: The inverse of the electrical resistivity of a solution containing electrolytes; measured via a wheatstone bridge and a standardized cell containing platinum electrodes, exactly 1 cm^2 in surface area, located a distance of 1 cm apart.

field capacity (specific retention): The amount of water retained by a soil (or aquifer) against the force of gravity.

fillable porosity: The volume of water that an unconfined aquifer stores during a unit rise in water table per unit surface area.

flux: The volume of water crossing a unit area of porous material per unit time.

gravitational head: The component of total hydraulic head relating to the position of a given mass of soil water relative to an arbitrary datum.

groundwater recharge: The flux of water across a water table.

groundwater zone: The hydrogeological region underlying a water table.

hydraulic conductivity: The proportionality factor in Darcy's equation; generally, a measure of the ease with which water moves through a porous system.

hydraulic gradient: The change in total hydraulic head of water per unit distance of flow.

infiltration capacity: The maximum rate at which water enters a soil.

isotropy: A porous system is isotropic with respect to hydraulic conductivity (or other properties) if values of the hydraulic conductivity are equal in all directions away from a point.

Klinkenberg effect: An effect which accounts for the greater permeability of a porous system to air movement than water movement; in particular, the velocity of air in the immediate vicinity of grains is finite because of slippage; in contrast, the velocity of water near grains is zero.

matric potential: The energy required to extract water from a soil against the capillary and adsorptive forces of the soil matrix.

matric suction: For an isothermal soil system, matric suction is the pressure difference across a membrane separating soil solution, in-place, from the same in bulk (Hillel, 1971).

matrix: The solid framework of a porous system.

osmotic potential: The component of the total soil-water potential associated with dissolved ions.

perched groundwater: A saturated groundwater body in the vadose zone, commonly developed at the interface between regions of varying texture.

percolation: The movement of water through the vadose zone, in contrast to infiltration at the land surface and recharge across a water table.

porosity: The fraction of a given volume of a porous system not occupied by solid grains.

pressure head: The head of water at a point in a porous system, negative for unsaturated systems, positive for saturated systems; quantitatively, it is the water pressure divided by the specific weight of water.

saturation extract: A solution extracted from a solids sample wetted with distilled water until a prescribed end point has been reached.

sodium adsorption ratio: A relationship between the concentration of sodium in an applied water or matrix solution and the concentrations of calcium and magnesium.

soil bulk density: The mass of dry soil in a volume of bulk soil.

soil peds: A soil structural unit, such as a crumb, prism, granule, or block.

soil-water characteristic curve: A graphical representation of the change in water content of a soil or porous media with changing soil-water pressure.

soil-water pressure: The pressure on the water in a soil-water system, as meaured by a piezometer for a saturated soil, or by a tensiometer for an unsaturated soil.

specific discharge: See flux.

specific yield: The volume of water that an unconfined aquifer releases from storage per unit surface area of aquifer per unit drop in the water table elevation.

tortuosity: A general term used to describe variations in the flow paths of water in a porous system caused by the presence of solid grains; quantitatively, it is the ratio of the distance travelled by a mass of water flowing through pores of a system to the length of the system (Hillel, 1971).

total hydraulic head: The sum of gravitational head and pressure head in water within a porous system.

total soil-water potential: The sum of the energy-related components of a soil-water system; i.e., the sum of the gravitational, matric, and osmotic components.

vadose zone: The hydrogeological region extending from the soil surface to the top of the principal water table.

water content: The amount of water stored within a porous matrix, expressed on either a volumetric (volume per unit volume) or mass (mass per unit mass) of solid.

water content profile: A log of the change in water content with depth in a profile through the vadose zone.